

Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK?

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Abstract

Purpose This study of seven foods assessed whether there are modes or locations of production that require significantly fewer inputs, and hence cause less pollution, than others. For example, would increasing imports of field-grown tomatoes from the Mediterranean reduce greenhouse gas (GHG) emissions by reducing the need for production in heated greenhouses in the UK, taking account of the additional transport emissions? Is meat production in the UK less polluting than the import of red meat from the southern hemisphere?

Methods We carried out a life-cycle inventory for each commodity, which quantified flows relating to life-cycle assessment (LCA) impact categories: primary energy use, acidification, eutrophication, abiotic resource use, pesticide use, land occupation and ozone depletion. The system boundary included all production inputs up to arrival at the retail distribution centre (RDC). The allocation of production burdens for meat products was on the basis of economic value. We evaluated indicator foods from which it is

possible to draw parallels for foods whose production follows a similar chain: tomatoes (greenhouse crops), strawberries (field-grown soft fruit), apples (stored for year-round supply or imported during spring and summer), potatoes (early season imports or long-stored UK produce), poultry and beef (imported from countries such as Brazil) and lamb (imported to balance domestic spring–autumn supply).

Results and discussion Total pre-farm gate global warming potential (GWP) of potatoes and beef were less for UK production than for production in the alternative country. Up to delivery to the RDC, total GWP were less for UK potatoes, beef and apples than for production elsewhere. Production of tomatoes and strawberries in Spain, poultry in Brazil and lamb in New Zealand produced less GWP than in the UK despite emissions that took place during transport. For foods produced with only small burdens of GWP, such as apples and strawberries, the burden from transport may be a large proportion of the total. For foods with inherently large GWP per tonne, such as meat products, burdens arising from transport may only be a small proportion of the total.

Conclusions When considering the GWP of food production, imports from countries where productivity is greater and/or where refrigerated storage requirement is less will lead to less total GWP than axiomatic preference for local produce. However, prioritising GWP may lead to increases in other environmental burdens, in particular leading to both greater demands on and decreasing quality of water resources.

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1 Introduction

Sustainable consumption and production aims to promote the use of goods and services with reduced environmental

impacts across their life cycles. Food production and consumption has been identified as a high impact category of consumption, when all life cycle stages are taken into account (Tukker et al. 2006). The increased global trade in food has led to a greater diversity of food chains supplying the UK consumer, many of which involve overseas production with long-distance transport to the UK, often under controlled temperature regimes (Mason et al. 2002). However, imports may provide advantages, due to climate, scales or types of production in other countries, in addition to price, consumer choice or year-round availability which Foster et al. (2006) suggested might be equivalent to an ‘ecological comparative advantage’. This raises questions regarding the comparative life-cycle burdens of different food supply chains and the extent to which some types of chains might be exporting our environmental burdens to countries outside the UK.

Williams et al. (2006) examined the environmental burdens and resource uses arising from the primary production of bread wheat, oilseed rape, potatoes, tomatoes, milk, eggs, poultry meat, beef, pig meat and sheep meat in Britain. Amongst the findings, the use of heated greenhouses for producing long-season tomatoes was clearly much more energy demanding, and greenhouse gas (GHG) emitting, than other crop production. Emissions and resource use in livestock production are about an order of magnitude greater per tonne of product delivered to the regional distribution centre (RDC) than from field crops as livestock are at a higher trophic level.

Foster et al. (2006) reviewed the evidence on life-cycle impacts of a range of commonly purchased fresh and processed foods in a UK ‘shopping basket’ and assessed the additional sustainability disadvantages of food miles. This study revealed a lack of good post-farm gate data applicable to UK conditions. Nevertheless, broad conclusions could be drawn regarding the scale of impacts along the supply chain of different food types in such a ‘shopping trolley’. In terms of primary energy, some of the most energy-intensive foods were shown to be livestock products and crops grown in heated greenhouses. A non-food comparison of roses imported from Kenya and the Netherlands found that carbon dioxide (CO₂) emissions were about six times less from Kenyan than Dutch roses, because the heating and lighting energy in the Netherlands greatly exceeded the airfreight emissions (Williams 2007).

There was a demand therefore to assess the different potential sources of foods to determine if there are modes or locations of production that offer significant reductions in energy and other resource use over others. For example, could there be significant energy reductions from increasing imports of field-grown tomatoes from the Mediterranean thus reducing the need for energy consumption in heated greenhouses in the UK, even when the additional transport

energy emissions are taken into account? Conversely, increased production of red meat in the UK from home-grown feeds could be more energy efficient and consume fewer resources than the import of red meat from the southern hemisphere. Further, consumers could be informed of the impact of their choices of produce.

Consideration also needed to be given to environmental burdens other than less energy-intensive production. For example, increased production of field-grown salads and vegetables in the Mediterranean area may damage water supplies, increase soil salinization and decrease water quality through eutrophication and pesticide contamination (Causapé et al. 2004). Hence, the assessment of the environmental burdens of producing food in the UK or elsewhere was carried out by means of a life-cycle assessment (LCA). The provision of comparative life-cycle burdens of alternative food supply chains for UK consumers will enable policy makers and consumers to understand the relative environmental impacts of consuming food obtained from other parts of the world compared with those same foods produced in the UK.

2 Methods

2.1 Scope and goals of the work

The goals and scope of this work, outlined below, were defined through the relevant ISO LCA standards: ISO 14040 and ISO 14044 (ISO 2006a; ISO 2006b). The models used are described in Section 2.3.

2.1.1 Goal of the study

The intended application was a comparative LCA of the environmental burdens and resource use arising from the production of seven key foods produced in the UK and elsewhere. A key requirement of this study was to make a comparison of out-of-season production in the UK against imports, but in some cases, we also made comparison between seasonal UK production and imports.

The reasons for the study were to assess whether there are modes or locations of production of the seven foods that offer significant reductions in GHG and other emissions and resource use over others.

The primary audience was Defra who funded the work, to provide policy-relevant information. Second were companies across the food chain, to identify the greatest sources of emissions, which could be addressed to reduce life-cycle burdens. Finally, consumers could be informed about the impact of their choices of out-of-season produce.

The study focused on current practice and a representative national average data for the reference year of 2005.

However, it also differentiated for seasonality with respect to crops like tomatoes and apples, but not for year-round production of meats.

We undertook sensitivity analysis focussing on key choices in production, storage and transport. For example, the comparison between greenhouses heated using gas vs. combined heat and power (CHP) or industrial waste heat. This was to illustrate the sensitivity to particular features of the stages in the chains and indicate where reductions in impacts might be possible. For reasons of brevity, those results are summarized only briefly in Section 3.1.

2.1.2 Scope of the study

The main relevant aspects are discussed in turn below.

2.1.3 Functions and functional units

The functional units for each commodity were 1 tonne of produce as delivered to the RDC and based on the final commodity sold to the consumer. The RDC represents both the retail sector and the equivalent for the hospitality trade. The exact products are given in Table 1.

While chicken may go into a number of other food products (from chicken nuggets to processed ready meals), we only assessed the primary product. For meat products, to allow a comparable LCA, products that might be substituted for each other were considered and used as the functional unit, i.e. frozen *chicken* breast and frozen hind beef cut.

Corrections to a standardised quality were not made as the foods considered (e.g. apples) are not sold at a standardised quality (e.g. sugar content).

2.1.4 Product system and system boundaries

The system boundary was set as the arrival of the produce at the RDC. The RDC was chosen as any differences between the commodities past this point (e.g. tomatoes may be eaten

raw or cooked) are the choices of the consumer, and we had no reason to consider steps post-RDC will not be common to a specific food irrespective of point of origin. This approach was also taken by Hospido et al. (2009). Hence, our results do not provide information on the total life-cycle burdens of the final product beyond the RDC, i.e. cooking and consumption. However, the comparisons we have made encompass any differences arising from the country in which the food was produced. Comparison of systems did include additional steps associated with e.g. out-of-season production.

The processes considered included all inputs and outputs directly associated with production: distribution; storage; refrigeration; transport; production and use of fuels, electricity and heat; use (and maintenance) of products; disposal of process wastes and products; recovery of used products; manufacture of ancillary materials; manufacture, maintenance and decommissioning of capital equipment (capital burdens); and additional operations (e.g. lighting and heating).

For some steps, rules were applied to investigate the likely proportion of emissions from those steps (materiality). Materiality was relevant for capital burdens (e.g. construction of greenhouses, construction of road transport vehicles or ships). The Cranfield model (Williams et al. 2006) includes most, but not all, capital burdens, dependent on available information and percentage impact of the capital burden on the total. To address other capital burdens, we estimated materiality using default data in the *Ecoinvent* database (<http://www.ecoinvent.org/database/>). The rules were that if the individual activity has burdens that are potentially >10 % of the process step (e.g. if the capital burdens from construction of road vehicles are >10 % of total transport burdens, or >1 % of the total burden), then they were included.

Billiard (2005) reported that for stationary as well as mobile refrigeration, on a life-cycle basis, ‘the emissions of refrigerant in a HFC-134a refrigerator represent only 1–2 % of the total contribution to global warming and emissions due to energy consumption represent 98–99 %’. This is because leakage of refrigerants is very small, and in consequence, despite the large global warming potential (GWP per molecule), our estimates of GWP from refrigeration and refrigerated transport are based only on the energy required for refrigeration.

2.1.5 Allocation procedures

Allocation was potentially important for those foods from which multiple products were derived. For example, the potential allocation of pre-farm burdens from cattle to leather as well as to meat. Hide is treated as a waste incurring no burdens in the Cranfield model since in the UK cattle are not slaughtered for their hides (Williams et al. 2006).

Table 1 The functional unit of each food

Commodity	Final commodity
Tomatoes	The main type is loose classic, but we also evaluated on-the-vine tomatoes, assuming that the weight excludes the attached vine, and cherry plum tomatoes
Strawberries	Whole fruits
Apples	Whole apples
Potatoes	Main and early potatoes
Chicken	Frozen chicken breast (i.e. weight as sold)
Beef	Frozen hind beef cut (i.e. weight as sold)
Lamb	Chilled, unless the NZ type is frozen

The allocation of food production burdens for different meat products was allocated on the basis of economic value, e.g. the allocation of poultry breast meat vs. the rest. This caused some potential problems with respect to the comparison of UK vs. (e.g.) Brazil because of the very different economic value of different parts of the animal in the two regions. An example of allocation by weight and economic value is provided in the Appendix ([Electronic Supplementary Material](#)).

2.1.6 Types of impact and methodology of impact assessment, and subsequent interpretation

We carried out a life-cycle inventory (LCI) for each commodity, and then progressed to a life-cycle impact assessment, associating inventory data with specific environmental impacts, for example, calculating GHG emissions, and using GWP to estimate the total global warming burden for each system. Each included assessment in five midpoint impact categories: global warming, acidification, eutrophication, abiotic resource depletion and ozone depletion following CML methodology (Guinée 2002), and four flow indicators: primary energy consumption (if possible, including the proportion as fossil energy), water consumption, land occupation and pesticide use. These were the impacts concluded to be most important in the discussions that took place during the development of the Cranfield LCA model (Williams et al. 2006). The impacts pre-farm gate were estimated using the Cranfield LCA model, and impacts post-farm gate were estimated using spreadsheets developed for this study based on data from the Ecoinvent database and NAEI (see Section 2.3). The pesticide rates for UK-grown feed crops were those reported in the UK's Pesticide Usage Surveys (e.g. Garthwaite et al. 2011). Those for soy were taken from the literature.

In addition, the following were assessed qualitatively by reference to the literature: loss of habitat and biodiversity, loss of soil C, sustainability of water supplies and pollution of watercourses or reserves. These impacts arise primarily as a result of production pre-farm gate.

2.1.7 Data requirements

The data requirements needed to cover differences in production, processing, storage and transport among the different localities of production and, in some cases, different seasons of production. The major focus of the study was on

- Energy inputs, raw material inputs, ancillary inputs, other physical inputs;
- Products;
- Emissions to air, water and soil resource use and impacts on ecosystem services.

While the baseline year was 2005 for agricultural systems, there is considerable year-to-year variability, and so a 5-year rolling average was used where data were available. The electricity generation burdens vary with the mix in each individual country. The LCI data on energy carriers, including UK and Spanish Electricity, came from the European Commission's 'International Reference Life Cycle Data System' (ILCD) core database (Wolf et al. 2012). For non-EU countries, LCI were constructed from public data of the International Energy Agency (<http://www.iea.org>), combined with energy carriers from the ILCD.

2.1.8 Model choices

One of the key factors was which set of GWP to use for comparing GHG emissions. There were three sets that could have been used: Inter-Governmental Panel on Climate Change (IPCC) 1995, IPCC 2001 and IPCC 2006. We used the GWP from the Third Assessment Report (IPCC 2006), which are also consistent with the requirements of PAS2050. The three IPCC approaches produce different values, particularly for agriculture where there are significant non-CO₂ emissions, i.e. from methane (CH₄) and nitrous oxide (N₂O). In addition, the emission of N₂O from N fixed by legumes was set to zero in the Third Assessment Report. PAS2050 is a guideline for product carbon footprinting (i.e. LCA with one objective). It makes specifications about methods, of which the specification for biogenic carbon is one. The UNFCCC is not primarily concerned with LCA, but the consistent reporting of GHG emissions across nations and economies. It is thus only relevant for this study for the purpose of providing values for GHG emission factors.

2.2 Choice of foods

We chose to evaluate a limited number of indicator foods from which we considered it possible to draw parallels for related foods whose production follows a similar chain.

2.2.1 Tomatoes, UK and Spain

Of the crops grown in heated greenhouses in the UK, tomatoes are grown in the largest amounts and have the greatest value. Hence, tomatoes were examined to identify the comparative environmental burdens of crops (e.g. cucumbers, lettuce and sweet peppers) grown in greenhouses in the UK. The example of imports from Spain illustrates the environmental burdens of production in other Mediterranean countries. Three types of tomatoes were examined: loose classic, vine cherry and plum. These represent a range of productivity (mass per unit area) in British heated houses that effectively determines the relative burdens between tomato types.

2.2.2 Strawberries, UK and Spain

Strawberries are produced in the UK and also imported in large quantities. Imports from Spain extend the season for strawberries. Although fruit of this type is considered particularly vulnerable to losses during transport, our data indicate that less than 5 % of strawberries are rejected on arrival at the RDC. The broad conclusions from this crop can also be applied to other soft fruits such as raspberries.

2.2.3 Potatoes, UK and Israel

Imports of potatoes from Israel and other Mediterranean countries are increasing, mainly to supply early potatoes when they are not available from UK sources. Considerable energy is required to transport such a bulky product. There are also implications for resource use in the countries of production, the widespread cultivation of potatoes for export may deplete water resources, increase soil salinization and reduce water quality in the countries of origin. Potatoes are vulnerable to considerable losses during storage and transport, especially if lifted under hot conditions and not cooled before transport. Our data indicate that c. 1 % of potatoes, either imported from Israel or grown in the UK, are rejected on arrival at the RDC, while wastage during storage is c. 1 % per month. The example of Israel may be applied for other Mediterranean countries.

2.2.4 Apples, UK and New Zealand

The trade in apples dominates that of other tree fruits. The cultivation of apples in those countries that supply the largest amounts of imports to the UK differs little to domestic production. However, many of those countries, New Zealand (NZ), South Africa and Chile, lie in the Southern hemisphere, and hence the energy needed to transport and refrigerate apples over large distances leads to an additional environmental impact. A main driver for imports from the southern hemisphere appears to be the sale of ‘fresh’ produce at a time when domestic produce would only be available from long-term storage. The example of apples illustrates the environmental consequences arising from import of other top fruit, e.g. pears.

2.2.5 Lamb, UK and New Zealand

Most NZ lamb is raised extensively with little N fertilizer or non-grass feed. Hence, notwithstanding the need for refrigerated transport over long distances, the environmental burden of this product might be considered relatively small. However, much lamb production in the UK is also extensive, and the whole industry is stratified vertically with hill, upland and lowland purebred flocks that are linked through

crossbred flocks to maximise hybrid vigour. Fertilizer N application rates on the grazed pastures increase as altitude decreases, but are still relatively small compared with dairy pastures. The supply of lamb is generally seasonal, and the well-established trade from NZ has been used to balance the spring–autumn supply from domestic production. Cold storage has allowed more lamb to become available outside the traditional supply seasons. This main term, together with transport from NZ and production differences, contributes to diverse burdens in the overall chain. The example of NZ lamb illustrates the environmental consequences arising from established meat production from grazing ruminants in countries under extensive production systems, e.g. beef from Argentina and Uruguay.

2.2.6 Poultry, UK and Brazil

In the years prior to this study, there was a very large increase in imports of poultry meat from Brazil and Thailand due to the increasing demand for inexpensive poultry meat within the EU. Since 1993, poultry imports to the UK have increased from c. 150 to c. 400 kt. Most poultry meat imported to Britain is deep frozen. Not only does the refrigerated transport of such large amounts of meat increase emissions but also there may be adverse environmental impacts within producing countries. As long as Brazilian maize and soya are price-competitive feeds, then some of the demand for poultry feed is likely to be met from Brazil, with the potential consequences for habitat destruction. The evaluation of this food chain therefore needs to take account not only of the costs of productions and transport of poultry but also that of soya and maize.

2.2.7 Beef, UK and Brazil

There has also been an increase in imports of beef from Brazil. Brazilian beef is predominantly raised by grazing, and there is concern that pastures may have been established on land that was formerly rainforest, albeit crops such as soya are likely to have been grown in the interval between clearance and pasture establishment. The example of these meat imports from Brazil illustrates the environmental consequences of increased meat production in countries with sensitive habitats vulnerable to encroachment by grazing.

2.3 Models used

All the pre-farm gate LCA calculations were implemented in Microsoft Excel, with some code in Visual Basic for Applications. Most LCI data on inputs were developed during project IS0205 from sources reported in Williams et al. (2006). Nitrate leaching from crops and long-term N mass balances were derived with SUNDIAL (Smith et al.

1997). The RothC simulation model (Coleman and Jenkinson 1996) was used to calculate soil C changes in Israel. Data of materials, such as plastics and metals, came from the *Ecoinvent* database (Goedkoop and Oele 2008). For the post-farm gate assessment, calculation spreadsheets were created specifically for the project. Emission factors (EF) were derived from a number of sources, mainly IPCC (2006), the UK National Air Emissions Inventory (<http://naei.defra.gov.uk/>) and the *Ecoinvent* database.

Calculations of GHG emissions from soils were largely based on the slightly modified IPCC (2006) Tier 2 inventory reporting guidelines, as interpreted by Williams et al. (2006). The main items included N₂O from N supplied as fertilizer, managed manure, grazing animal N excretion and crop residue returns together with secondary emissions from leached nitrate (NO₃) and volatilised ammonia (NH₃). The same EF were used for all countries. Soil emissions from polytunnels were assumed to behave in the same way as in open ground. Nitrous oxide emissions from recirculating nutrient solutions were calculated as by Williams et al. (2006). Emissions of C (as CO₂) from soil [or indeed sequestration in soil organic carbon (SOC)] were generally not included on the basis that most of the systems being studied are likely to be in or close to steady state. In two cases, where a steady state was not assumed, these were treated separately, i.e. potatoes in Israel and beef in Brazil.

We addressed whether the CO₂ taken up by the tomato crop should be debited from the total CO₂ emitted by burning natural gas. We decided that in common with other crops that take up CO₂ and emit O₂, the CO₂ taken into non-woody plant biomass is in a transient store. Since the bulk of the biomass will be in the fruits that are eaten and used as a basis for respiration, we did not debit the CO₂ taken up by the crop that had been emitted by burning natural gas. This approach is consistent with that set out in the final draft of the Carbon Trust and Defra sponsored Publicly Available Specification (PAS) 2050—specification for the assessment of the life cycle GHG emissions of goods and services.

2.4 Data sources

Full details of data inputs and sources are supplied in the final project report (AEA et al. 2008). The yields and carcass weights used in the estimates and other key input data are provided in Tables 2 and 3, respectively, at the end of this section.

2.4.1 Tomatoes

The data used were averages of UK production. Values for energy use included a dataset supplied by the Farm Energy Centre obtained from a large number of growers. Information on Spanish production systems were derived from

Table 2 Yields and carcass weights used to estimate emissions from production, transport, processing and packaging to the RDC

	Type	UK	Country	non-UK
Tomatoes	Classic loose, t/ha	541	Spain	120
	Classic vine, t/ha	227		79
	Baby plum, t/ha	196		59
Strawberries	Whole fruits, t/ha	19	Spain	39
Potatoes				
Main	ware, t/ha	42	Israel	45
Early	ware, t/ha	22		26
Apples	Whole fruits, t/ha	26	New Zealand	63
Poultry	Average slaughter weight, kg	1.7	Brazil	2.4
Beef	Average carcass weight, kg	610	Brazil	460
Lamb	Average carcass weight, kg	19.4	New Zealand	17.0

literature values and data from one large producer considered typical for the region. Being one example of each tomato, they cannot be assumed to be national averages.

The main differences between UK and Spanish tomato production were due to the much greater requirement for heating in the UK, to provide an optimal growing environment, versus the longer transport distances for imports from Spain. All fully commercial UK production is under glass, while in Spain most production for the UK market is in polyethylene-covered houses of varying sophistication. Nearly all UK production is based on hydroponic systems using either substrates, such as Rockwool, or nutrient film systems, whereas Spanish systems tend to be more varied, and soil is still widely used as the growth medium. In the UK, greenhouses are mainly heated with natural gas, either by stand-alone boilers or CHP units.

For CHP, GHG emissions were estimated by considering the gas used in the house and calculating a credit for the exported electricity, based on a comparison with the most modern generating method. The CHP system that we considered was a unit that burns gas, e.g. a reciprocating gas engine of 1 MW (suitable for 1 ha) generating capacity. The internal heat and electricity needs of the greenhouse are met by the CHP unit, and the CO₂ is used for tomato production as though from a stand-alone boiler. Most of the generated electricity is surplus and hence exported to the national grid. Comparison was between generation by CHP in which tomatoes are a co-product and conventional electricity generation for the national grid. A combined cycle gas turbine (CCGT; Defra 2008) was used for the assessment, representing a comparison of new marginal generating capacity rather than the existing generating mix.

A few sites have been developed where local, industrial waste heat and CO₂ are used. The CO₂ from combustion is fed

Table 3 Input data used to estimate emissions arising post-farm gate: journey distances and payloads, processing inputs, energy requirements for cooling and freezing

		UK	% Non-UK
Tomatoes	Distance to RDC, km	205	2,300
	Lorry payload, to processing plant, t	8.0	8.0
	Lorry payload, to RDC, t	21.0	22.5
	% Backhaul	70	70
	Tones required for 1 t to reach RDC	1.1	1.2
	Electricity used for processing, kWh/t	84	47
	Wastage from processing and packing, %	3	13
	Type of heat source	CHP	35
	Waste heat and CO ₂	13	
	Natural gas	52	
Strawberries	Distance to RDC, km	208	2,300
	Lorry payload, to processing plant, t	8.0	8.0
	Lorry payload, to RDC, t	21.0	22.5
	% Backhaul	70	70
	Tones required for 1 t to reach RDC	1.0	1.0
	Energy used for initial cooling, MJ/t	76	76
Apples	Wastage, %	<1	1
	Distance to RDC, km	220	18,340
	Lorry payload, to processing plant, t	8.0	8.0
	Lorry payload, to RDC, t	21.0	na
Potatoes	% Backhaul	70	70
	Tones required for 1 t to reach RDC	1.2	1.02
	Electricity used for washing etc., kWh/t	76	76
	Electricity used for storage etc., kWh/t/day	0.54	na ^a
	Distance to RDC, km	220	4,200
	Lorry payload, to processing plant, t	8.0	8.0
	Lorry payload, to RDC, t	21.0	na
	% Backhaul	70	70
Potatoes	Wastage, at processing plant, %	30	15
	Wastage, from processing plant to RDC, %	1	15
	Electricity used in processing plant, kWh/t	55	55

Table 3 (continued)

		UK	% Non-UK	
Lamb	Distance to abattoir/cutting/packing plant, km	85	75	
	Lorry payload, to abattoir, t	21	21	
	% Backhaul	0	0	
	Distance from abattoir to port, km	na	400	
	Lorry payload, to port, t	na	21	
	% Backhaul	na	0	
	Distance to RDC, t	200	18,340	
	% Backhaul	70	70	
	Electricity, slaughter and processing, kWh/head	19	19	
	Heat, slaughter and processing, kWh/head	4.5	4.5	
	Ratio of carcass weight to dead weight	0.46	0.42	
	Ratio of lamb at RDC to live weight	0.48	0.36	
Beef	Tonnes of cattle required for 1 t beef at RDC	2.1	2.8	
	Distance to abattoir/cutting/packing plant, km	185	150	
	Lorry payload, to abattoir, t	21	80	
	% Backhaul	0	0	
Beef	Distance from abattoir to port, km	na	400	
	Lorry payload, to port, t	na	80	
	% Backhaul	0	0	
	Distance to RDC, t	200	9,293	
	% Backhaul	70	70	
	Electricity, slaughter and processing, kWh/head	25.5	25.5	
	Heat, slaughter and processing, kWh/head	31.0	31.0	
	Ratio of carcass weight to dead weight	0.53	0.53	
	Ratio of beef at RDC to carcass weight	0.70	0.70	
	Tonnes of cattle required for 1 t beef at RDC	2.7	2.7	
	Poultry	Distance to abattoir/cutting/packing plant, km	34	50
		Lorry payload, to abattoir, t	21	21
% Backhaul		0	0	

Table 3 (continued)

	UK	% Non-UK
Distance from abattoir to port, km	na	500
Lorry payload, to port, t	na	21
% Backhaul	na	70
Distance from abattoir to RDC, km	200	9,293
% Backhaul	70	70
Electricity, slaughter and processing, kWh/head	0.44	0.44
Heat, slaughter and processing, kWh/head	0.22	0.22
Ratio of carcase weight to dead weight	0.65	0.65
Ratio of chicken at RDC to carcase weight	0.80	0.80
Tonnes of chicken required for 1 t meat at RDC	1.9	1.9
Shipping EF Shipping alone	Energy, MJ/t/km	0.24
Including chilling		0.29
Including freezing		0.27
Shipping alone	CO ₂ emission, kg/t/km	0.0178
Including chilling		0.0215
Including freezing		0.0200

^a Included with shipping emissions

into greenhouses to enhance photosynthesis and crop productivity. This increased yield will slightly reduce the energy input per tonne of product. There are clear cost advantages to this as well as environmental ones; hence, more such developments may follow. A main constraint is the availability of suitably flat land close to sources of waste heat and CO₂. Breweries and distilleries are possible sources.

Pesticide use is greater in Spain, partly because disease pressure is greater over the winter months, and the typical level of Spanish greenhouse technology restricts the use of cultural control techniques used in the UK. The greater threat from insect-vectored viruses and limited development of integrated pest control strategies also results in additional pesticide application in Spain.

The typical Spanish export season is November to June and the UK production from March to October. Hence, the Spanish season generally complements UK production. The assessment presented here relates to the principal long-season production system with April–October fruit production. Winter production in Northern Europe is still limited,

and the bulk of UK supply at this time of year comes from Spain, the Canaries, Morocco, Italy and Israel.

2.4.2 Strawberries

Data from the University of Hertfordshire study of UK strawberry production (Warner et al. 2010), obtained from growers and covering a range of production systems, were used to analyse and model UK strawberry production. The Spanish system modelled was based on average data coupled with some specific values from the literature.

We estimated that 75 % of the total UK crop was grown under protection, and this proportion was taken for the weighted mean of production. Because strawberry runners are normally propagated outside and with little fertilizer inputs, it was reasonable to conclude that the burdens would be much less than the main production phase of the business, so this part has been excluded, but would be broadly the same between the UK and Spain.

UK strawberry production supplies the market between April and November. This requires the use of season extension techniques, such as cropping under glass, ever-bearing varieties to extend the cropping season and also selective planting of June-bearers so that they first crop before June. However, it has not been possible to develop economically viable production systems to produce large volumes of UK fruit from the autumn through to the spring, and during this period, the market is normally supplied with imported fruit from countries including Morocco, Spain, Egypt, Israel and the USA.

2.4.3 Potatoes

Data for UK potato production from the Cranfield LCI project IS0205 (Williams et al. 2006) were updated with new energy sources and made compliant with IPCC 2006. The main sources for Israeli production were a local producer, the scientific literature, the internet and the Israeli Ministry of Agriculture. About 80 % of Israeli potato production is in the Negev region and 15 % in Sharon area of central Israel. We had enough information about systematic differences between these two areas provided by the Israeli sources on fertilizer inputs and irrigation use to model them separately and produce weighted average results.

Production in Israel and the UK uses essentially the same equipment and practices for crop establishment, production and harvesting. However, Israeli crops are typically produced on very sandy soils (much is reclaimed desert), and these crops have much greater water demand than comparable UK crops. Furthermore, the irrigation water for UK crops is obtained from a range of local sources (boreholes, surface water and reservoirs), whereas water in Israel is either supplied from boreholes or by canal or pipe from

more distant sources. Much of the UK main crop is stored, whereas storage is limited in Israel, and most of the exports to the UK are supplied as fresh product, allowing for initial cooling and transit.

Production in Israel is divided between autumn and spring planting. Early potatoes are planted from late September to early October and harvested from late November through to April. Winter main crops may be planted around the same time, but grow for longer, and are harvested from March. Only main crop types are grown in spring (January–February) for a May–June harvest. Israeli potatoes supplied to the UK generally complement UK production. The appropriate UK storage times for a comparison with Israeli main crop potatoes are from harvest in autumn until the Israeli March crop arrives and from harvest in autumn until the May–June crops arrive from Israel. There is an interval after harvest in Israel to allow some initial chilling, packing and transit to the UK. The storage of summer-consumed early potatoes is limited because the tubers have limited storability and the bulk of this product (both UK and Israeli) is supplied directly post-lifting.

For maincrop potatoes, a model was applied of production followed by storage. Efficiency tends to reduce as the store is emptied, at least where stores are little more than large cooled warehouse. However, if the store comprises segregated units, efficiency will not necessarily decrease as the store is emptied. Energy input is greatest (c. 12 kWh/tonne/week, as electrical energy is not primary) in the first week of storage when energy is needed to cool the potatoes to the desired storage temperature as well as to maintain the difference with the warm ambient temperature. Once the potatoes have been cooled, and the outside temperature decreases, the energy input also decreases to c. 4.5 kWh/tonne/week in January. Thereafter, the energy requirement increases, as the ambient temperature gets warmer. The result is for storage to increase the energy requirement for potatoes non-linearly and nearly to double burdens by July following harvest in October.

2.4.4 Apples

The main data sources for apple production in NZ were the NZ Pipfruit Monitoring report (MAF 2006a), Saunders et al. (2006), Milà i Canals et al. (2006) and Milà i Canals (2003). The NZ Pipfruit reports provide yield and activity data based on commercial orchards. While these are secondary data, they are in the public domain and appear to be the closest there is to official national average yields. UK data sources included industry information from the project team, Defra statistics on yields and crop areas and the Pesticide Usage Survey.

A general apple production model was produced in which the basic operations are almost the same between countries. Some exceptions being that pruning and picking

in the UK are done without hydraulic platforms (due to UK production being based on dwarf root stocks, albeit this difference is diminishing as NZ production is also moving to dwarf root stocks), NZ irrigation uses electricity for pumping rather than diesel. NZ uses more renewable energy in its generating mix, so that the CO₂ equivalent per kilowatt hour is less than in the UK, but oil imports have bigger overheads (c. 1.06 the UK values). This model assumes an average apple composition for both countries. All burdens are greatly influenced by yield as exemplified by primary energy use (PEU) and GWP. Estimates of typical NZ yields ranged from 56 to 70 tonne/ha (MAF 2006a) and were greater than those reported for the UK (23 to 38 tonne/ha). Estimates of pre-farm gate (agricultural production) burdens were made using all four yield estimates, and the means for UK (30.5 tonne/ha) and NZ (63 tonne/ha) production were taken and reported here. These differences in yield dwarf differences in actual production systems, and hence we can be reasonably confident that primary production burdens are smaller in NZ than in the UK. There have been substantial investments in new UK orchards over recent years, and significant increases in average yield would be expected as a result. The change in UK productivity is likely to have a beneficial impact on the various measures detailed in this report, but we were not able to fully assess those.

With respect to the need for storage, we took September (UK) and March (NZ) as the median harvest months. We then compared UK produce that had been stored for 5 months with NZ apples arriving in the UK without prior storage in NZ and hence marketed fresh. Since the proportion of NZ produce stored must increase up to the 3 months until July when the first UK crops are ready, we also compared un-stored UK produce with apples stored NZ for 3 months.

2.4.5 Lamb

Lamb production in NZ and the UK is broadly similar with grazing the main source of nutrition in both countries. Pastures in NZ are generally more productive for longer periods of the year than in the UK (e.g. NZ receives more solar radiation) and rely on less input of synthetic N fertilizer and more use of clover. This results in negligible housing of NZ sheep, albeit sheep are only housed for a relatively small period in the UK. NZ flocks also have lesser inputs of winter (or dry season) forage and concentrates. Although Saunders et al. (2006) implied that neither are used, the Sheep and Beef Monitoring of MAF (2006b) records purchases of both forages and concentrates for sheep, and they were included in the more detailed analysis of Barber and Lucock (2006).

The UK national flock structure is complex. That in NZ is somewhat simpler, but still has some movement of, for

example, store lambs from hill to lowland farms to be finished, albeit such transhumance is less pronounced in NZ. The structural model integrates lamb meat outputs from hill, upland and lowland into a weighted average tonne of lamb. These could be analysed separately, but it was not a main focus of this project, which was to compare production in different countries.

The ratio of carcase weight to liveweight differs between the UK and NZ, carcase weight being 54 % and 47 %, respectively. This difference is likely to have arisen from breeding programmes giving more emphasis to meat production for UK sheep compared with the NZ breeds. Further wastage of 5 % of the carcase was reported during the cutting of carcasses into joints, chops etc.

The husbandry of sheep is an enterprise for which allocation of burdens is particularly important. Sheep are raised for both meat and wool, and the relative importance of those products differs between the UK and NZ. Lamb meat is the main output in both countries, but mutton from the slaughtered ewes is an inevitable co-product that is nutritionally sound, but out of fashion in the UK. The NZ flock produces more wool per ewe than UK sheep (5 and 3 kg, respectively, annually). Economic valuation is the most rational way of allocating the burdens among disparate co-products and was used here in pre-farm gate assessment. Different data sources were examined, and both MAF and the Meat and Wool New Zealand Economic Service provided data on the relative value of carcasses and the price of wool per kilogram in NZ. The English Beef and Lamb Executive record data on UK lamb meat prices and live cull ewe prices, and the wool price is publically accessible. Some judgement was needed to resolve disparate data, and it was assumed that wool is worth more in NZ as are mutton carcasses. This led to using allocations of all burdens of 64 % and 74 % for NZ and the UK, respectively.

2.4.6 Poultry

Brazilian (export) and UK poultry meat production use essentially the same strains of birds, and as long as diets are correctly formulated and there are no disease outbreaks, then bird performance should be the same. We assumed that manure is managed equally well between the countries and that crop responses are of equal value per unit manure N. Most poultry production is in the southern parts of Brazil where heat stress is not a systemic problem. Some heating is needed for young birds.

2.4.7 Beef

Modelling the LCA of beef production from Brazil was more complex than UK production. Most exported beef comes from single suckle, extensively reared, Nelore (*Bos*

indicus) cattle. The long-lived and resilient Nelore breed is generally raised on Cerrado pastures with a small proportion finished in feedlots. Part of the motive for using feedlots is that the grazing season is split into wet and dry seasons and feedlots offer a way of finishing cattle, particularly in the dry season. Pastures receive very little N fertilizer and stocking rates are much less than in the UK. Winter feeding is restricted mainly to a urea and mineral supplements rather than the forage and concentrates that are used in the UK. This means stock gain weight in the wet season and may lose weight in the dry season. Energy use is least in Brazil on the Pampa in which winter supplements do not appear to be necessary. Energy input increases in the Cerrado as pasture renewal rate goes up. The feedlot finishing system is about twice as energy intensive as on the Cerrado. Several assumptions needed to be made about Brazilian beef production owing the great difficulty in obtaining reliable activity data. This uncertainty should be taken into account in the interpretation of our results.

We modelled Brazilian beef as it operates to supply the EU export trade. There are other states of Brazil with different types of land and on which different management may be practised. We did not model all production systems.

2.4.8 Shipping

Few data are available on the energy requirement needed for the long-distance transport of produce by ship, and the consequent GHG emissions. This may be because transport by ship is very energy efficient, with estimates of between 10 and 70 g CO₂/tonne/km, compared with estimates of 20–120 and 80–250 g CO₂/tonne/km for rail and road, respectively. Nevertheless, the quoted range is large, and over the distances from the southern hemisphere to the UK of up to 20,000 km, use of inappropriate EF could lead to significant errors. In addition, the produce under consideration is transported under refrigeration, whether chilled or frozen, which requires additional energy, which is not covered by the standard energy use and emission data. Estimates of this additional burden range from 10 % to 50 % of basic energy use.

Many foods are transported to the UK in reefer ships. The basic energy requirement of such ships was estimated to be 0.24 MJ/tonne/km. This increases to 0.27 MJ/tonne/km when transporting frozen produce and 0.29 MJ/tonne/km for the transport of chilled. More energy is needed for ventilation and cooling of chilled products, which are more active than frozen products, even if the thermal losses are lower than those of frozen products (Buhaug et al. 2008). Based on these data, a default EF for CO₂ emissions of 0.0178 kg/tonne/km was used in the estimates of GHG emissions from shipping. Account was also taken of extra energy inputs and consequent emissions for refrigeration and also the embodied emissions in building the ship. This

is an area where simplifying assumptions, e.g. using a standard inventory value per tonne per kilometre, may not be valid. It is an area in which more activity data is highly desirable, especially as these values are of great interest both to parties in the UK and overseas.

2.5 Uncertainty

All scientific measurements and models contain some uncertainty. There are very large uncertainties associated with N₂O emissions and large ones for enteric methane (CH₄). The way in which these are aggregated in LCA (or indeed carbon footprinting alone or many other environmental assessments) tends to reduce the errors, but this may not always be the case. Aggregation may also compound errors. In consequence, the way in which these should be treated in a comparison is still the subject of debate. This is not simply an experimental comparison in which two normal distributions may be compared using a conventional statistical test. Fully defined uncertainties will not generally apply owing to, for example,

- Large uncertainties associated with the emission factors (e.g. N₂O from soils), which contribute to high uncertainty in one part of an estimate, will mean the estimate has a high uncertainty. However, such emission factors are present in both systems under comparison and therefore should not affect that comparison.
- Variances being very hard to estimate, e.g. where an absence of data requires expert opinion to ascribe a value or where an input has no declared variance. Note that many standard emission factors do not as yet have publically accessible, defined associated variances.

In common with other reported studies (e.g. Vergé et al. 2008; Virtanen et al. 2011), our assessment was not based entirely on data collected solely or originally for this study. Hence, it was not possible to explicitly define the uncertainty of all components of the GHG budget presented in this assessment. Lloyd and Ries (2007) noted that ‘Additional research is needed to understand the relative importance of different types of parameter, scenario, and model uncertainty and to determine whether guidelines regarding the types of uncertainty and variability that should be included in LCA can be established’.

In consequence, we offer no conclusive statements about statistical significance between the results reported here. As a guide, in other similar system models, the uncertainty (quantified as the coefficient of variance—standard deviation/mean) may lie in the region of 15 % to 35 % for factors like NH₃ emissions (Van Gijlswijk et al. 2004; Webb and Misselbrook 2004), but possibly over 50 % for N₂O (IPCC 2006). Given this and some understanding of the other factors, it seems likely that some criteria are

significantly different up to the farm gate. These, with caution, are thought to include energy, acidification, land occupation, pesticide use, abiotic resource use and volatile organic carbon. All the text that follows should, however, be read with qualification that calculated statistical significance is not implied in any statement.

3 Results

The results reported below are presented in Table 4.

3.1 Tomatoes

Per tonne of loose tomatoes delivered to the RDC, PEU and GWP of UK produce were four and three times greater, respectively, than for Spanish produce (see Table 4). The energy required for transport from Spain was moderate at c. 3.6 GJ/t compared with totals of 9.6 and 36.2 GJ/t for Spanish and UK production, respectively (see Table 4). Abiotic resource use was c. 35 % greater for UK production, mainly due to energy consumption and the greater resources needed to build the permanent greenhouses. In contrast, the eutrophication and acidification potentials of Spanish produce were both c. twice as large as those of UK produce. Pesticide use was seven times greater for Spanish production, while water use was 50 % greater. UK yields are greater per unit area, and hence only 20 % of the land is needed.

Our estimate of 2.24 t CO₂ per tonne for UK production is similar to the estimates reported by Biel et al. (2006) for greenhouse production in Sweden (2.72 t CO₂ per tonne), at the wholesalers. Our estimate would be expected to be smaller than earlier ones due to increase in energy efficiency from practices such as CHP.

The differences in energy use and GHG emissions between UK and Spain are least for baby plum tomatoes because they are grown with more energy and in more sophisticated houses in Spain than are loose classic tomatoes.

The assessment also compared UK tomatoes from CHP heated greenhouses against Spanish tomatoes (Fig. 1). The benefits of CHP in reducing energy use and GHG emissions vary from about 32 % if substituting for CCGT to 91 % if substituting for the current mains mixture. The benefits are greater for the lesser-yielding baby plum tomatoes because the proportion of burdens attributable to energy is proportionately larger than for loose classic. There is clearly substantial uncertainty about quantifying the benefits of CHP, so the estimates presented here should be regarded as an illustrative estimate rather than a definitive statement. The improvements from using waste heat and CO₂ are larger than from using CHP (when compared with CCGT). Energy use and GHG emissions pre-farm gate were reduced by over 90 % for all three types of tomatoes. In consequence, while

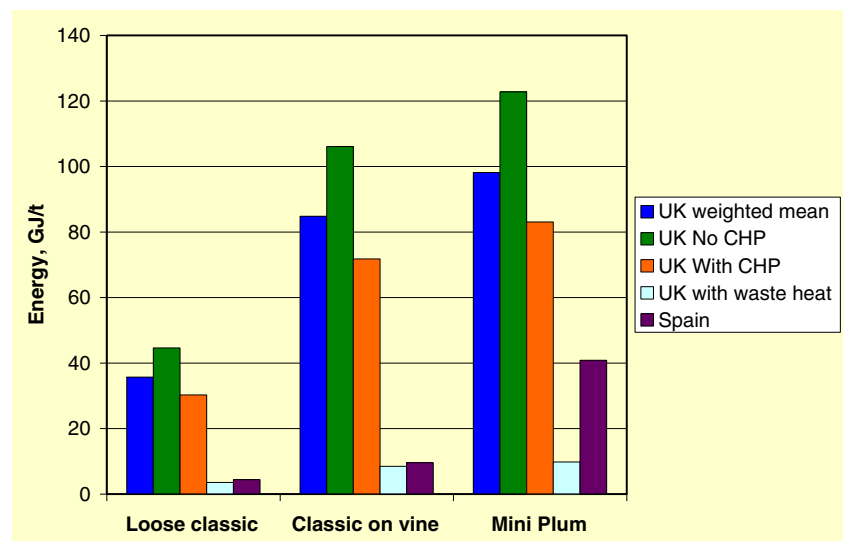
Table 4 Total primary energy use (PEU) and global warming potential (GWP) and amounts due to import to the UK, abiotic resource use, eutrophication, acidification potentials, water use, land requirement and pesticide use total emissions per tonne of product

	PEU ^a GJ/t		GWP t CO ₂ eqv./t		Abiotic resource use, kg Sb eqv.		Eutrophication potential, kg PO ₄ eqv.		Acidification potential, kg SO ₂ eqv.		Water use, m ³		Land requirement, m ²		Pesticide use, dose/ha	
	UK	non-UK	UK	non-UK	UK	non-UK	UK	non-UK	UK	non-UK	UK	non-UK	UK	non-UK	UK	non-UK
Tomatoes (loose classic), Spain																
Pre-farm gate total	34.1	4.4	2.1	0.3	16.7	10.3	0.1	0.1	1.4	1.7	24	36	18.5	89.1	0.3	2.2
Transport to the UK	NA	3.6	NA	0.3	18.2	13.5	0.2	0.5	2.4	4.1	24	36	18.5	89.1	0.3	2.2
Total	36.2	9.6	2.2	0.7												
Strawberries, Spain																
Pre-farm gate total	12.9	8.3	0.8	0.3	14.9	7.2	2.6	3.4	7.7	7.1	108	128	0.054	0.026	1.1	0.4
Transport to the UK	NA	3.0	NA	0.3												
Total	14.6	13.3	1.0	0.9	0.7	0.9	1.9	1.0	0.5	0.8	24	100	0.04	0.03	0.6	0.4
Early potatoes, Israel																
Pre-farm gate total	1.5	1.9	0.2	0.2												
Transport to the UK	NA	8.6	NA	0.3	14.5	12.0	20.6	20.6	41.8	39.0	15	15	0.6	0.4	NA	NA
Total	2.8	10.5	0.3	0.5	19.7	18.9	20.8	39.4	42.9	160.7	15	15	0.6	0.4	NA	NA
Apples, New Zealand																
Pre-farm gate total	2.1	1.2	0.2	0.1	16.7	10.3	0.1	0.1	1.4	1.7	24	36	18.5	89.1	0.3	2.2
Transport to the UK	NA	7.5	NA	0.7	18.2	13.5	0.2	0.5	2.4	4.1	24	36	18.5	89.1	0.3	2.2
Total	5.1	11.2	0.4	0.9												
Lamb, New Zealand																
Pre-farm gate total	17.9	12.2	13.5	9.7	14.9	7.2	2.6	3.4	7.7	7.1	108	128	0.054	0.026	1.1	0.4
Transport to the UK	NA	7.5	NA	0.6												
Total	30.7	37.0	14.1	11.6	0.7	0.9	1.9	1.0	0.5	0.8	24	100	0.04	0.03	0.6	0.4
Poultry, Brazil																
Pre-farm gate total	15.9	12.2	2.5	2.1	48.8	5.7	129.4	81.2	229.1	39.5	800	800	0.9	25.7	NA	NA
Transport to the UK	NA	4.1	NA	0.4	53.7	11.0	129.5	83.0	230.0	51.1	800	800	0.9	25.7	NA	NA
Total	21.1	24.7	2.8	2.6												
Beef, Brazil																
Pre-farm gate total	41.3	7.8	23.8	31.7	1.8	5.5	0.4	3.7	1.8	24.1	10	88	0.04	0.02	0.6	0.3
Transport to the UK	NA	4.0	NA	0.3												
Total	44.4	17.1	24.0	32.2	20.3	13.2	84.1	60.7	57.6	51.6	125	125	9	5	NA	NA

The unit is derived as follows: if a pesticide is applied at 0.45 of its full dose per hectare, then this equates to 0.45 dose hectare. This is analogous to the tonne-kilometre used in transport LCI data
NA not applicable

^a PEU includes non-renewable and renewable sources. Details of the proportions are in Appendix 1 of AEA et al. (2008)

Fig. 1 Energy use for tomato production in the UK with and without combined heat and power (CHP) and Spain



total GWP from using CHP would still be three to five times greater for UK production than Spanish, the use of waste heat could lead to UK produce of less GWP than that from Spain (see Fig. 1).

There is a clear trend in UK tomato production to move away from stand-alone boiler heating and CO₂ supply, and it seems most probable that some older sites with poor thermal performance and stand-alone boilers will be replaced. Cost is clearly one driver, but better management of ventilation with computer control and the use of technologies like thermal screens have helped improve the thermal performance of greenhouses. This should be no surprise as improving technical efficiency is a key to reducing burdens.

Most newer UK installations also harvest rainwater and store runoff in reservoirs (a number of older sites have also been modified to increase rainwater capture). This reduces the load on mains water and boreholes, which is particularly important, as water resources in the drier South and East have become increasingly stretched.

There are also trends for convergence, which complicate drawing conclusions from any one study. UK heated greenhouse production has become more energy efficient while Spanish production has become more energy intensive in order to respond to demands for a greater variety and quality of produce.

3.2 Strawberries

The PEU and GWP per tonne of strawberries delivered to the RDC were c. 9 % and 11 % less, respectively, for Spanish produce than for UK. Pre-farm gate PEU and GWP were c. 65 % and 40 %, respectively, those of UK production, due primarily to yields in Spain being about twice those of the UK. Post-farm gate (i.e. after produce has left the farm) PEU and GWP were three and four times

greater for Spanish produce than for UK. The PEU and GWP associated with transport from Spain accounted for only c. 22 % and 30 % of the totals for Spanish produce, respectively.

As a consequence of the greater energy input, the acidification potential of UK produce was c. 9 % greater than that of Spain. Land requirement and abiotic resource use were also greater for UK production each by a factor of 2. Abiotic resource use in Spain was less than for UK production due to the lesser infrastructure requirement. However, N utilization efficiency is generally less in Spain, leaving greater N residues in autumn, and hence leaching is greater than in the UK, leading to Spanish production having 30 % greater eutrophication potential than UK produce.

While imported strawberries from Spain appear to have smaller PEU and GWP burdens than UK production, the substantial use of irrigation water has potentially led to water supply and contamination concerns in Spain, and there may be increases in energy use and GWP as a result of future growth in water demand. Further work on these issues would be useful before specific comparisons are made between the two systems.

3.3 Potatoes

Per tonne of early potatoes delivered to the RDC, the PEU and GWP of early potatoes from Israel were c. 4 and 1.7 times greater than for UK produce (see Table 4). This is due to the greater energy intensity associated with irrigation (water supply), smaller yields (compared with UK main crop) and greater transport. Eutrophication and acidification potentials (see Table 4), abiotic resource and water use were all greater for Israeli production, although pesticide and land use were less (see Table 4).

Produce from Israel is available before the traditional UK early crop when domestic produce is largely maincrop potatoes that have been stored. The storage of maincrop potatoes in the UK may require as much energy to ensure a year-round supply as the initial production. We therefore carried out a comparison of main crop production in the UK and Israel and estimated the impact of different storage periods on emissions from UK production. The total PEU and GWP were c. 60 % and 40 % less, respectively, than Israeli production (without storage), which supplies main crop potatoes in early–mid summer.

These results reflect the importance of yield when assessing emissions per tonne of produce. We estimate 0.9 GJ/t and 0.11 t CO₂ equivalent per tonne for UK maincrop potatoes but between 1.3 and 1.9 GJ/t PEU and 0.16 to 0.21 t CO₂ equivalent per tonne of lesser yielding UK and Israeli earlies. Our estimate of 0.9 GJ/t primary energy used in the pre-farm gate production of maincrop potatoes in the UK is rather less than the earlier estimates of 1.3 and 1.4 GJ/t, respectively, reported by Williams et al. (2006) and Pimentel and Pimentel (1996), but storage was included by Williams et al. (2006), and in this study, storage is accounted for post-farm gate. Our estimate of GWP for maincrop potatoes pre-farm gate at 0.11 t CO₂ equivalent per tonne is less than the 0.16 t CO₂ equivalent per tonne estimated by Lillywhite et al. (2007). The Lillywhite et al. (2007) estimates of acidification and eutrophication potential of maincrop potato cultivation were also c. 45 % greater than the estimates reported here.

Our post-farm gate estimates for transport energy requirement at 335 MJ/t are greater than those of Matsson and Wallen (2003), who reported 200 and 600 MJ/t for packaging, compared with our estimate of 67 MJ/t. However, their data were for peeled potatoes, ours are for whole and packaging burdens would be expected to be greater for peeled potatoes. Foster et al. (2006) reported that cooling and storage account for 40 % of PEU in potato production. This depends upon the time of harvest (harvesting early in September under UK conditions will require cooling, whereas lifting in October does not) and length of storage. In this study, storage and cooling accounted for between 26 % and 43 %, depending upon the length of storage.

3.4 Apples

Due to the energy required for transport to the UK, total PEU and GWP were 2.2 and 3.1 times greater for apples produced in NZ than in the UK. Eutrophication potential and acidification potential were both much greater for NZ produce (see Table 4). These greater emissions also arose largely from shipping. Default EFs for SO₂ and NO_x (kilograms per kilogram fuel) for heavy fuel oil, the main fuel source for reefer ships, are much greater than for most

other transport fuels. This is due not only to differences in the composition of fuels but also to emission limits for road transport.

Post-farm gate GWP from NZ production was dominated by transport to the UK (80 %), albeit some of those emissions arose from the need to chill the apples during transport. For UK produce storage (29 %), packaging (26 %) and processing (23 %) were of similar importance. Storage length made little difference to the balance of emissions.

Our estimates of PEU for pre-farm gate apple production were 1.2 and 2.1 GJ/t, respectively, for NZ and UK. Milà i Canals et al. (2006) reported PEU for NZ apple production in the range 0.4 to 0.7 and 0.4 to 2.0 GJ/t in Europe. Blanke and Burdick (2005) calculated PEU of 2.8 GJ/t in both Germany and NZ. Saunders et al. (2006) calculated primary production energies of 0.95 and 3.0 GJ/t for NZ and the UK, respectively. Saunders et al. (2006) assumed 6 months storage for UK apples, so incurring a further 2.1 GJ/t from refrigeration. This seemed to ignore the substantial part of the UK crop that is eaten without storage (apart from at the RDC).

Our estimates of energy requirements for packaging, transport within the country of origin and for storage within the UK (compared with Germany) are similar to those of Blanke and Burdick (2005) who did not appear to report the energy required for processing within the pack house. However, they reported only 2.8 GJ/tonne PEU to be required for shipping apples from NZ to Germany based on a smaller estimate of energy use for shipping of 0.11 MJ/tonne/km, less than half of the one we used. In addition, the extra energy burden from cooling in this study was c. 20 %, whereas Blanke and Burdick (2005) used only c. 12 %. These differences led to our estimate of PEU required for shipping to be more than double that of Blanke and Burdick at 7.4 GJ/tonne. Mason et al. (2002) reported a CO₂ EF for shipping of 0.007 kg/tonne/km. Our data indicate that for reefer ships, the EF is about 2.5 times greater (0.018 kg/t/km), and should be increased by a further 20 % to account for refrigeration energy. Moreover, in this study, we took account of the energy required for, and emissions from, ship manufacture and recovery and transport of fuel oil. Hence, our estimate of CO₂ emissions from shipping, at 612 kg CO₂ equivalent per tonne is much greater than that quoted by Mason et al. (2002). As well as the large difference in EF used, Mason et al. (2002) also assumed 100 % re-loading efficiency; we took a value of 70 %. Again, as with our estimate of PEU required for shipping, we consider our estimate of GWP to be soundly based.

3.5 Lamb

The main burdens of PEU and GHG emission pre-farm gate are about 30 % less in NZ than the UK (see Table 4). The energy value for NZ accords with the range found by Barber and Lucock (2006) of 11 to 16 GJ/t carcass. Both sets of results are greater than those of Saunders et al. (2006) who

found 8.6 GJ/t carcass, but the full Agriculture Research Group on Sustainability data of Barber and Lucock (2006) were not available to them.

Our estimate of total PEU was c. 21 % greater for NZ produce, but total GWP was c. 20 % less for NZ. The estimate of PEU required to transport the NZ lamb to the UK, 7.5 GJ/tonne is c. seven times greater than the estimate of Saunders et al. (2006), while our estimate of GHG emissions is also larger than they reported. However, our estimate is similar to that cited by Carlsson-Kanyama et al. (2003) of c. 6 GJ/tonne more for lamb imported from ‘overseas’ than for lamb produced in Sweden. Our estimate of the energy required by shipping (c. 0.285 MJ/tonne/km) is greater than that quoted by Weidema et al. (1995) for ‘overseas’ shipping of 0.2 MJ/tonne/km. However, those values were for fuel oil consumption and do not appear to include energy embodied in the ship’s manufacture or refrigeration.

We report a much greater PEU, per tonne of meat delivered to the RDC, from the slaughter and processing of lambs than of cattle. The difference arises primarily from the requirement for electricity for lamb slaughter and processing which, at 19.0 kWh per head, is only c. 25 % less than that required for cattle (25.5 kWh per head) even though cattle may weigh up to 15 times more. However, the processing of lambs is more energy intensive.

The largest source of GWP pre-farm gate was enteric fermentation in both the UK and NZ. The greater GWP arising from grazing in the UK was due to the greater amounts of fertilizer N applied to UK pastures. The large energy input needed to process lamb, in comparison with other meats, lead to processing being the largest source of GWP post-farm gate in both countries. Transport, either to the UK or within the UK, gave rise to most of the rest of the GWP arising post-farm gate. The transport of lamb from NZ gives rise to large emissions of potentially acidifying and eutrophying gases. As noted above, these arise from the greater emissions of NO_x and SO₂ (by two orders of magnitude) from fuels used in shipping compared with fuels used in road transport.

Abiotic resource use and land requirement were about 50 % and about 80 % greater for UK production due to the greater land required for less productive pastures in the UK and a requirement, for some flocks, of housing during lambing. Since pesticides are not applied to sheep pastures in either country and requirements for water are the same, there were no differences in the consumption of those resources.

3.6 Poultry

The burdens of primary production are broadly similar between Brazil and the UK, but energy use is c. 25 % less in Brazil and GWP c. 17 % less (see Table 4). There are three main reasons for this. First, a main feed, soya, is

locally produced and has much smaller transport burdens in Brazil. Second, Brazilian poultry houses are essentially naturally ventilated, but with limited fan use in very hot weather, and less heating fuel is used for young birds. Third, structures are simpler so that the housing burdens are smaller. The lesser energy requirement of broiler production in Brazil was also noted by Biel et al. (2006).

Total PEU from the production of poultry in Brazil, at 24.7 GJ/t, was c. 18 % greater than that for the UK (20.9 GJ/t). However, GWP at 2.6 t CO₂ equivalent per tonne was c. 9 % less than the UK estimate (2.8 t CO₂ equivalent per tonne; see Table 4) due to the majority of electricity in Brazil being generated from renewable resources. Our estimates for GWP are similar in total to those of Foster et al. (2006) of 3.1–3.3 t CO₂ equivalent per tonne for chicken meat ‘ex-slaughterhouse’. Our calculation of GWP for transport to the RDC was only 0.025 t CO₂ equivalent per tonne.

In both countries, c. 80 % of pre-farm gate GWP arose from the production of feeds. Most of the remainder were a result of gaseous emissions from the buildings. Meat processing accounted for most post-farm gate GWP in the UK, but only 20 % of the total in Brazil, again due to electricity generation from renewable resources in Brazil. Our results show that for UK poultry, the greatest burdens post-farm gate, c. 85 % of the totals, were from processing, with c. 9 % arising from transport. Emissions from packaging were generally <1 % of the post-farm gate total. Processing of poultry in Brazil accounted for 65 % of PEU, but only 20 % of GWP, with 80 % of GWP arising from transport to the UK.

While pre-farm gate impacts on eutrophication and acidification were similar from production in the UK and Brazil when transport emissions were taken into account, total eutrophication and acidification potentials were c. two and four times greater from production in Brazil (see Table 4). This was due to the emission of SO₂ and NO_x from the combustion of marine fuel oils. Other impacts were similar since production in both countries is mainly in large units, in which the birds are reared indoors thus generating similar requirements for land, water and abiotic inputs.

3.7 Beef

Emissions of GHGs are greater from the Brazilian systems mainly because of greater enteric CH₄, reflecting the relatively slow growth and reproductive rates of Brazilian cattle. Another difference is that about 30 % of the beef calves finished in the UK originate from the dairy sector. Because a dairy cow needs to produce a calf in order to produce milk, this overhead is carried by the dairy sector, consequently reducing breeding overheads in the UK.

The estimate for the UK of 16 t CO₂ equivalent per tonne for the whole carcass to the farm gate is somewhat greater than those of Casey and Holden (2006) of 11–13 t CO₂

equivalent per tonne for Ireland. In contrast with the conclusions of Smith et al. (2005), we did not find that extensive beef production lead to smaller GHG emissions per tonne than intensive. That finding appears to have been derived from the cited results of Casey and Holden (2006) who calculated emissions from three different systems in Ireland. Those differences were not large (11–13 t equivalent per tonne), and the differences between organic and conventional production in a single country are unlikely to have been as great as the differences between the systems evaluated here. In particular, given the very favourable climate for year-round grass production in Ireland, it is unlikely the time taken to reach maturity by the organic livestock was any greater than the time taken by the more intensively reared animals. It was this slower growth rate on the poorer-quality pastures in Brazil that led to greater GHG emissions per tonne of beef.

In both the UK and Brazil, enteric fermentation is the largest source of GWP from pre-farm gate production. However, in the UK, substantial emissions also arise from the production of forages and the maintenance of grazed pastures, primarily from the manufacture of fertilizers and from subsequent emissions of N₂O. Our estimates of the PEU required for beef processing, at c. 1.8 GJ/t for the UK and c. 4.1 GJ/t in Brazil were within the range of results quoted in earlier work (1.6 GJ/t cited by Smith et al. 2005; 4 GJ/t, Foster et al. 2006).

The assessment presented is an attributional LCA, not a consequential LCA. It thus does not take into account the effects of increasing production at the margins. This is clearly a sensitive matter in Brazil because much marginal production in the country is associated with the destruction of native rainforests. While most, if not all, production for export to the EU is outside the Amazonian area, the consequences of increasing supply to the EU could be to displace production for the domestic market (and other exports) further into rainforest. It was beyond the scope of this study to quantify any such effects, but their existence must be acknowledged.

Acidification (*5) and eutrophication (+50 %) were much greater from rearing beef in the UK than in Brazil. This was due to the much greater use of fertilizer in the UK. Abiotic resource use was c. eight times greater from production in the UK, since beef cattle are typically housed indoors for 5–6 months of the year requiring infrastructure such as buildings, yards etc. In Brazil, beef are reared almost exclusively outdoors. In contrast, the land requirement from extensive production in Brazil is around 30 times greater than in the UK. Requirements for water and pesticides are similar in both countries.

4 Discussion

A key driver for the import of foods to the UK is consumer demand (Carlsson-Kanyama 1998). Mila i Canals (2006)

reported that over 70 % of consumers in UK urban areas consider that they should have the choice of purchasing any food product at any time of the year. Until the latter part of the twentieth century, the UK diet in winter included less fresh fruit and vegetables, but more dried, bottled, canned and frozen produce, than is now common. All such preservation methods incur burdens and are attended by losses of quality and quantity. Some normally added sugar to fruits or salt to vegetables, which may have made them more attractive to some consumers, but was unlikely to have conferred as great health benefits as eating fresh produce (albeit the provision of calories as sugar may have been less deleterious to a population whose calorie intake was less, and physical activity greater, than is normal today). Greater consumption of fresh, or fresher, produce is potentially associated with health improvements through improved diet (if healthier choices are made by individuals). Hence, if there is a demand for all-year availability of produce, and it is deemed that this demand should be met, then the most resource-efficient means of meeting such a demand may be to import recently harvested produce rather than store UK produce for long periods. This may remain so even when that produce has to travel very long distances.

In general, our findings were that for foods, GHG emissions are not usually directly related to PEU, but also due to emissions of N₂O from N fertilizer and manure applications and CH₄ from enteric fermentation in the case of livestock production. Both have a GWP considerably greater than that of CO₂ by *296 and *25, respectively (IPCC 2006).

Where yields are substantially less than the alternative, but overall inputs are comparable, emissions per tonne of produce will be greater: e.g. apples and strawberries in the UK, beef in Brazil. Therefore, sourcing from productive areas rather than trying to boost yields with large energy inputs, whether directly in the case of heated greenhouse production, or indirectly through greater inputs of N fertilizer, is a potential means of reducing emissions per tonne of produce. This appears to the case even when there is greater transport distance associated with sourcing, as transport emissions are generally a small proportion of production, albeit our estimates of PEU and GWP from shipping are greater than in some previous studies. However, in cases where there are similar emissions per tonne of produce, then longer distance transport will be more important component of the relative environmental burdens between sources.

It can be seen that for some products, in particular apples produced in NZ, the GWP arising from transport to the UK significantly alters the balance between UK and overseas production. However, this is partly a reflection of the small GWP arising from apple production. For foods such as lamb, with a much greater GWP burden per tonne of production, the additional emissions from transport do not cancel out the advantage from production in NZ.

Overall, our findings generally agree with other authors in some areas, e.g. primary production and cold storage, but the more contentious differences appear in shipping.

Another crucial factor is the need for refrigeration. This appears less significant for meat products, which are always likely to need cooling wherever produced, although the length of journey has an impact. Refrigeration is a greater factor for those fruits and vegetables, which can only be harvested over a limited period and will therefore need storage, under conditions which inhibit deterioration, if they are to supply consumers for much or all of the year. For example, while UK potatoes, due to greater yields and less need for water and transport, have advantages over Mediterranean produce, this advantage can be partly or maybe wholly lost by the need for prolonged storage. Hence, it may be more effective to import seasonal produce that can be sold without prolonged storage than to keep the UK produce for 6–8 months. The exact balance depends very much on production burdens and losses during storage. It also needs to be remembered that long shelf-life products, such as frozen products, can be transported slower, and therefore, in a more efficient manner.

For tomatoes and strawberries, packaging was one of the main contributors to post-farm gate GWP, being c. 70 % in both cases for the UK and 25–40 % for Spain. The proportion was smaller for post-farm gate burdens from apple production (10–25 %) and <10 % for potatoes. With respect to these horticultural crops, post-farm gate GWP was relatively small at <1 t CO₂ equivalent per tonne of produce. Packaging led to a smaller proportion of emissions associated with meat production, comprising <1 % of emissions from poultry, both UK and Brazil, 1–3 % of burdens for lamb and from <1 % (GWP) to 13 % for beef.

The main focus of this study was to calculate the PEU and consequent GHG emissions from the production of selected foods. Another objective was to assess whether the production of food outside the UK, albeit with reduced GHG emissions, might be exporting other environmental problems to those countries. Our results suggest this may be the case. The lesser consumption of PEU and smaller GWP from producing tomatoes and strawberries in Spain has to be balanced against the greater adverse impacts on water use and quality in that country. The increase in agricultural production in parts of Spain has led to aquifers becoming polluted by nitrate and other salts, to a lowering of the water table, and increasing energy use in water supply (Causapé et al. 2004; Schofield et al. 2001). Torrellas et al. (2012) concluded that reduced fertilizer use would be the most cost-effective means of reducing the environmental impact of tomato production in SE Spain. Providing water for Spanish horticulture is an increasing national concern (Gonzalez and Romero 1991). Desalinization, either of seawater in coastal areas or of polluted aquifers elsewhere, may

become necessary, though these changes must be seen against the backdrop of cross-sectoral water demand increases and socio-economic change. Any demand for desalinization will be driven by multiple pressures and should not be attributed solely to horticulture. However, desalinization or water pumping increases the energy demand and burdens of growing crops. Water will be a growing problem this century, as populations and demand grow. Hence, some production systems might be considered ‘better’ overall even if emitting more GHGs.

Increased production in those countries may also be increasing pressure in biodiversity. However, we did not find evidence of particular impacts of production on biodiversity, soil, water resources or water quality in NZ.

5 Conclusions

Despite the additional environmental burdens arising from long-distance transport, we found that for some foods, the overall environmental impact was less from production outside the UK. Our results are consistent with the concept of ‘ecological comparative advantage’ suggested by Foster et al. (2006). It seems certain that, unless the UK embarks on a radical change of lifestyle, from drastically reducing the consumption of livestock products to embracing veganism, and moving to a more seasonal diet, food will need to be imported from overseas to meet the demand. Since a substantial volume of imports appears to be inevitable, it may be better, when considering the GWP of food production, to accept that imports from countries where productivity is greater and/or where refrigerated storage requirement is less will lead to less total GWP than axiomatic preference for local produce. However, prioritising GWP may lead to increases in other environmental burdens, in particular leading to both greater demands on and decreasing quality of water resources. These results broadly support the conclusions of Foster et al. (2006) that ‘there is little evidence that local supply and consumption reduces the overall environmental impact of food production’.

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References

AEA, Cranfield University, Moorhouse E, Paul Watkiss Associates, AHDBMS, Marintek (2008) Comparative life cycle assessment of

- food commodities procured for UK consumption through a diversity of supply chains. Final report for Defra project FO0103, 18 pp and Appendices. <http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=15001&FromSearch=Y&Publisher=1&SearchText=FO0103&SortString=ProjectCode&SortOrder=Asc&Paging=10#Description>
- Barber A, Luccock D (2006) Total energy indicators: benchmarking organic, integrated and conventional sheep and beef farms. Research report number 06/07, ARGOS. Downloaded from www.argos.org.nz
- Biel A, Bergström K, Carlsson-Kanyama A, Fuentes C, Grankvist G, Lagerberg Fogelberg C, Shanahan H, Solér C (2006) Environmental information in the food supply system. Report number, ISRN FOI-R-1903-SE. Swedish Defence Research Agency, Defence Analysis, Stockholm, 117 pp
- Billiard (2005) Refrigerating equipment, energy efficiency and refrigerants. <http://www.iifir.org/en/doc/1059.pdf>
- Blanke MM, Burdick B (2005) Food (miles) for thought: energy balance for locally-grown versus imported apple fruit. *Environ Sci Poll Res* 12:125–127
- Buhaug Ø, Corbett JJ, Endresen Ø, Eyring V, Faber J, Hanayama S, Lee DS, Lee D, Lindstad H, Mjelde A, Pålsson C, Wanqing W, Winebrake JJ, Yoshida K (2008) Updated study on greenhouse gas emissions from ships: phase I report. International Maritime Organization (IMO), London
- Carlsson-Kanyama A, Ekström MP, Shanahan H (2003) Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. *Ecol Econ* 44:293–307
- Casey JW, Holden NM (2006) Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. *J Environ Qual* 35:231–239
- Causapé J, Quílez D, Aragüés R (2004) Salt and nitrate concentrations in the surface waters of the CR-V irrigation district (Bardenas I, Spain): diagnosis and prescriptions for reducing off-site contamination. *J Hydrol* 295:87–100
- Coleman K, Jenkinson DS (1996) RothC-26.3—a model for the turnover of carbon in soil. In: Powlson DS, Smith P, Smith JU (eds) Evaluation of soil organic matter models using existing, long-term datasets. NATO ASI Series I, vol 38. Springer, Berlin, pp 237–246
- Defra (2008) Guidelines to Defra's greenhouse gas conversion factors for company reporting. June 2008. Accessed from: <http://www.defra.gov.uk/environment/business/envrp/pdf/ghg-cf-guidelines2008.pdf>
- Foster C, Green K, Bleda M, Dewick P, Evans B, Flynn A, Mylan J (2006) Environmental impacts of food production and consumption: a report to the Department of Environment, Food and Rural Affairs. Manchester Business School, Defra
- Garthwaite DG, Thomas MR, Parrish G, Smith L, Barker I (2011) Pesticide usage survey report 224. Arable crops in Great Britain 2008. <http://www.fera.defra.gov.uk/plants/pesticideUsage/documents/arable2008.pdf>
- Goedkoop M, Oele M (2008) Introduction to LCA with SimaPro 7. Pre Consultant, Amersfoort, <http://www.pre-sustainability.com/simapro-lca-software>
- Gonzalez A, Romero E (1991) Influence of pumps on salinity of a coastal aquifer with legal protection (a zone in the north of Isla-Cristina, Huelva, Spain). *Water Sci Technol* 24:251–260
- Guinée JB, Gorée M, Heijungs R, Huppes G, Kleijn R, Koning A de, Oers L van, Wegener Sleeswijk A, Suh S, Udo de Haes HA, Bruijn H de, Duin R van, Huijbregts MAJ (2002) Handbook on life cycle assessment. Operational guide to the ISO standards. Kluwer Academic Publishers, Dordrecht, The Netherlands
- Hospido A, Milá I, Canals L, McLaren S, Truninger M, Edward-Jones G, Clift R (2009) The role of seasonality in lettuce consumption: a case study of environmental and social aspects. *Int J Life Cycle Assess* 14:381–391
- IPCC (2006) Draft 2006 IPCC guidelines for national greenhouse gas inventories
- ISO (2006a) ISO 14040: Environmental management—Life cycle assessment—Principles and framework
- ISO (2006b) ISO 14040: Environmental management—Life cycle assessment
- Lillywhite R, Chandler D, Grant W, Lewis K, Firth C, Schmutz U, Halpin D (2007) Environmental footprint and sustainability of horticulture (including potatoes)—a comparison with other agricultural sectors. Final report of Defra project WQ0101
- Lloyd SM, Ries R (2007) Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. *J Industr Ecol* 11:161–179
- MAF (2006b) Sheep and beef monitoring report. A short-term financial and physical forecast reflecting farmer and industry perceptions of farming figures, trends and issues. July 2006. Downloaded from: www.maf.govt.nz/
- MAF (Ministry of Agriculture and Food) (2006a) Pipfruit monitoring report a short-term financial and physical forecast reflecting grower and industry perceptions of pipfruit figures, trends and issues in Hawkes Bay and Nelson. April 2006 MAF. <http://www.maf.govt.nz/mafnet/rural-nz/statistics-and-forecasts/farm-monitoring/>
- Mason R, Simons D, Peckham C, Wakeman T (2002) Life cycle modelling CO₂ emissions for lettuce, apples and cherries. Report for Department of Transport. Executive summary available at: http://www.dft.gov.uk/pgr/freight/research/coll_lifecyclemodellingco2emissi/lifecyclemodellingco2emissio3226
- Matsson B, Wallen E (2003) Environmental LCA of organic potatoes. In: Proceedings of the 26th International Horticultural Congress, ISHS. *Acta Hort* 619:427–435
- Milà i Canals L (2003) Contributions to LCA methodology for agricultural systems. Site dependency and soil degradation impact assessment. PhD Thesis, Universitat Autònoma de Barcelona, Barcelona
- Milà i Canals L, Burnip GM, Cowell SJ (2006) Evaluation of the environmental impacts of apple production using life cycle assessment (LCA): case study in New Zealand. *Agric Ecosys Environ* 114:226–238
- Pimentel D, Pimentel M (1996) Food energy and society. CRC Press Taylor & Francis Group Boca Raton, FL, USA
- Saunders C, Barber A, Taylor G (2006) Food miles—comparative energy/emissions performance of New Zealand's agriculture industry. Research Report No. 285. Lincoln University, Canterbury
- Schofield R, Thomas DSG, Kirkby MJ (2001) Causal processes of soil salinization in Tunisia, Spain and Hungary. *Land Degrad Devel* 12:163–181
- Smith J, Bradbury N, Glendinning M, Smith P (1997) Application to SUNDIAL to simulate nitrogen turnover in crop rotations. *Quant Appr Syst Anal* 10:87–92
- Smith A, Watkiss P, Tweedle G, McKinnon A, Browne M, Hunt A, Treleven C, Naish C, Cross S (2005) The validity of food miles as an indicator of sustainable development. Report for Defra. <http://statistics.defra.gov.uk/esg/reports/foodmiles>
- Torrellas M, Antón A, López JC, Baeza EJ, Pérez Parra J, Muñoz P, Montero JI (2012) LCA of a tomato crop in a multi-tunnel greenhouse in Almería. *Int J Life Cycle Assess* 17:863–875
- Tukker A, Huppes G, Guinée J, Heijungs R, de Koning A, van Oers L, Suh S, Geerken T, Van Holderbeke M, Jansen B, Nielsen P, Eder P, Delgado L (2006) Environmental impact of products (EIPRO). Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. European Commission Joint Research Centre, EUR 22284 EN, 136 pp. http://ec.europa.eu/environment/ipp/pdf/eipro_report.pdf

- van Gijlswijk R, Coenen P, Pulles T, van der Sluijs J (2004) Uncertainty assessment of NO_x, SO₂ and NH₃ emissions in the Netherlands. Rapport nr. R 2004/100. TNO Environment, Energy and Process Innovation, Apeldoorn
- Vergé XPC, Dyer JA, Desjardins RL, Worth D (2008) Greenhouse gas emissions from the Canadian beef industry. *Agr Syst* 98:126–134
- Virtanen Y, Kurppa S, Saarinen M, Katajajuuri J-M, Usva K, Mäenpää I, Mäkelä J, Grönroos J, Nissinen A (2011) Carbon footprint of food—approaches from national input-output statistics and a LCA of a food portion. *J Cleaner Prod* 19:1849–1856
- Warner DJ, Davies M, Hipps N, Osborne N, Tzilivakis J, Lewis KA (2010) Greenhouse gas emissions and energy use in UK-grown short-day strawberry (*Fragaria xananassa* Duch) crops. *J Agric Sci* 148:667–681
- Webb J, Misselbrook TH (2004) A mass-flow model of ammonia emissions from UK livestock production. *Atmos Environ* 38:2163–2176
- Weidema BP, Pedersen RL, Drivsholm TS (1995) Life cycle screening of food products: two examples and some methodological proposals. Akademiet for de Tekniske Videnskaber, Lyngby, 193 pp
- Williams AG (2007) Comparative study of cut roses for the British market produced in Kenya and the Netherlands. Cranfield University, Cranfield, Report for World Flowers
- Williams AG, Audsley E, Sandars DL (2006) Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Cranfield University, Bedford
- Wolf M-A, Pant R, Chomkamsri K, Sala S, Pennington D (2012) The International reference life cycle data system (ILCD) handbook. JRC reference reports. Publications Office of the European Union, Luxembourg, 65 pp