PARALLEL SESSIONS 2

2B / LCA and Footprinting

Regional water footprint and water management: the case of Madrid region (Spain)

José Soler-Rovira[1,*], Juan Manuel Arroyo-Sanz[1], Hugo Conde-Marcos[1], Carlos Sanz-Zudaire[1], Alfredo Mesa-Moreno[1], Sergio Gil-Pascual[1]

[1]Department of Agronomy, Technical Agricultural Engineering School, Technical University of Madrid, Ciudad Universitaria s/n, 28040-Madrid, Spain

ABSTRACT

Water resources and water footprint of the production and consumption in Madrid region were estimated, considering blue water (water resources), green water (soil moisture), grey water (polluted water) and virtual water (water trade in products imported and exported in the region). Water resources in Madrid relay mainly in surface waters and rainfall, so the periodic occurrence of meteorological droughts implies the scarcity of water supply. The main users of blue water are households, municipalities and agriculture. Production water footprint is approximately 2,000 hm^3 per year, almost four times the blue water available resources in the region. Consumption water footprint is around 9,000 hm^3 per year from which 80% is net imported virtual water, so Madrid depends on external water resources in order to supply the actual consumption needs. These should be more sustainable reconsidering the consumption pattern, as 60% of the water footprint is due to meat production.

Keywords: blue water, food, green water, grey water, water resources.

1. Introduction

Water is an essential element in the environment but also for economic activity and human development. The study of water resources and water use has been done elsewhere (e.g. Alcolea and García-Alvarado, 2006; Sotelo, 2006). These studies include the water resources and water cycle (blue water) in a city or region. They also may include water quality and treatment or water prices and markets, but they do not consider the use of green water (effective rainfall) or the virtual water content of products consumed in that region. Virtual water concept was introduced by Allan (1998); and it is the water required to produce a certain product. This concept was used in practical applications (e.g. Chapagain and Hoekstra, 2003), considering the water footprint as the blue water withdrawal (water from rivers, wells and reservoirs) and the net virtual water imported. Chapagain and Hoekstra (2004) introduced in the calculations the green water (soil moisture) used for crops production in a certain region. In order to include the impact or water pollution, the grey water is considered the water volume required to dilute pollutants to as such extent that this water would fulfil water quality standards (Chapagain *et al.*, 2006). Hoekstra and Chapagain (2008) presented the evolution of water footprint and virtual water concepts into one coherent framework.

In a high-density populated area as Madrid region water is a key element. Water resources and water use has been studied elsewhere (Alcolea and García-Alvarado, 2006; Sotelo, 2006). Water demand is driven by urban supply (households and municipality uses), which is increasing due to population growth and changes in societal lifestyles (e.g. private gardens

* Corresponding Author. e-mail: jose.soler@upm.es

and swimming pools, public parks, sport facilities and amusement parks). However, this water footprint does not take into account virtual water of consumption goods and services.

The production water footprint of a region is the water used for households, industrial and agricultural purposes in that region regardless of where the products are actually consumed (Hoekstra *et al.*, 2009). This footprint is made up of blue, green and grey water. But the water footprint of a region is the total volume of water used globally to produce the goods and services consumed by its inhabitants. The total water footprint is made up of two components: internal and external. The internal water footprint is the volume of water needed to grow and provide the goods and services which are produced and consumed inside that region. The external component results from consumption of imported goods, so it is water used for the production in the exporting region.

The aim of this work is to study the water footprint of the production and the consumption in Madrid region (Spain), and to evaluate the implication for water management in that territory.

2. Methods

Water footprint of Madrid region was studied. Madrid is located in the centre of Iberian Peninsula (40 ° N, 3° W). Its surface area is circa 8,000 km^2 with an average altitude of 650 m, with a mountain zone with summits over 2,000 m high. The climate is Mediterranean semiarid, with an average rainfall of 436 mm and temperature of 14.6 °C in Madrid City and 1,326 mm and 6.4 °C in the mountains. It is a highly populated region with circa 5.5 million inhabitants and 744 inhabitants per km^2, mainly dedicated to the tertiary sector (77% of gross added value of the region) as it is the capital of the country.

Water footprint was estimated considering water resources, production and consumption water footprint. Resources of surface and ground water were obtained from literature and government statistics (Alcolea and García-Alvarado, 2006; Naredo *et al.*, 2009; Instituto de Estadística, 2010; INE, 2010d). Water evaporation from reservoirs was computed with the reference evapotranspiration (ETo) calculated with the Hargreaves method (Villalobos *et al.*, 2002) with local climatic data and an evaporation coefficient (Doorenbos and Pruitt, 1979). Net evaporation was computed considering reservoirs surface (Instituto de Estadística, 2010) and rainfall.

Average statistical data were used for the years 2000 to 2005, when they were available; if not, accessible and most representative data were used. Years 1971-2000 climatic series data were used for rainfall and ETo calculations. All results were calculated in a yearly basis.

Green water was estimated as the rainfall water stored in cultivated soils and evapotranspirated by crops. It was calculated for rainfed and irrigated crops, fallow land, pastures, meadows and parks and green areas. Crops evapotranspiration and effective rainfall were computed with CROPWAT programme (FAO, 2010a) using site specific climatic data and crop coefficients (MAPA, 2002; Villalobos *et al.*, 2002). Green water in rainfed crops was calculated as the evapotranspired effective rainfall stored in soils. For irrigated crops it was computed as the difference of crop evapotranspiration and calculated irrigation water requirements. In fallow land a yearly water balance was done considering an evaporation coefficient as a function of rainfall frequency. Cultivated areas were taken from MARM (2008).

Blue water was considered as the fresh water withdrawn from water bodies that was used and not returned. Blue water withdrawals were computed from government statistics: households and municipalities (Instituto de Estadística, 2010), industry (INE, 2010d) and agriculture (INE, 2010c). Livestock water consumption was estimated with the number of animal heads (INE, 2010a) and the average consumption by each species (MIMAM, 2007). Losses from the water network were taken from Instituto de Estadística (2010).

Grey water was calculated as the volume of water required to dilute pollutants to as such extent that water quality reaches acceptable standards. Nitrate leaching was calculated for cultivated land with average fertilization rates per crop (MAPA, 2004) and leaching fraction from literature. Grey water from wastewater treatment was estimated for the total volume of treated wastewater (Instituto de Estadística, 2010), wastewater nitrogen composition (INE, 2010b) and N limits in Wastewater Directive. Dilution of deposition of nitrogen species in reservoirs was estimated with deposition rates of oxidized and reduced nitrogen (EMEP, 2010), surface or reservoirs (Intituto de Estadística, 2010) and drinking water quality standards for nitrate and ammonium.

Water footprint was computed considering consumption (MAPA, 2007), production (MARM, 2008) and trade of food (Instituto de Estadística, 2010) in the region and the virtual water content of crop and livestock products (Chapagain and Hoekstra, 2004). Water footprint of animal feed was computed with data of livestock units in the region (INE, 2010a), feed consumption (FAO, 2010b) and virtual water content of crop and livestock products (Chapagain and Hoekstra, 2004). Industrial products footprint was estimated with trade data of Madrid region (Instituto de Estadística, 2010), water use in industry (INE, 2010d) and industry production (INE, 2010e).

3. Results and discussion

The structure of water resources in Madrid region is described in table 1. Despite the large water resources located in the tertiary aquifer, the available resources relay mainly on surface waters, and hence in rainfall. Then, the periodic occurrence of meteorological droughts implies water supply scarcity. The soil moisture (green water) is also important since its value almost double the blue water resources.

Table 1: Water resources in Madrid region (hm^3).

Natural resources		**Infrastructures**		**Available resources**	
Rainfall	4,195	Reservoir capacity	1,154	Water stored in reservoirs	515
Tertiary aquifer	3,000,000	Pumping capacity	106	Pumped water	87
				Soil moisture	1,119
		Total	1,260	Total	1,721

There is a high proportion of green water withdrawn by rainfed crops, as they cover the 88% of the cultivated land in the region (table 2). This water is mainly used in pastures and meadows, cereal crops and olive trees (table 3). It is noticeable the green water consumption in parks and gardens, that equals that of irrigated crops.

Table 2: Water withdrawals in Madrid region (hm^3).

Green water		**Blue water**	
Irrigated crops	54	Households	343
Rainfed crops	1,007	Agriculture	172
Gardens and parks	58	Other	82
		Municipalities	49
		Industry	49
		Livestock	2
		Network losses	70
Total	1,119	Total	697

Blue water is mainly used by households and municipalities (urban use) and agriculture (table 2). Some field crops (maize, barley, lucerne, etc.) require a big amount of blue water and they are irrigated by gravity, which means low application efficiency. Important volumes of water are also used to irrigate vegetables (melon, lettuce). The water footprint of production activities in Madrid region (table 4) shows green, blue and grey water components. Green water use is three times higher than blue water. It is also remarkable the water required to dilute the pollutants in order to agree to quality standards (grey water). In this grey component wastewater is the most important factor, due to the necessary water to dilute the nitrogen not removed in wastewater treatment plants. Agriculture and deposition grey water show similar values. The total production water footprint adds up around 2,000 hm^3 per year, almost four times the blue water available resources in the region.

Table 3: Water use in agriculture (hm^3).

Green water		Blue water	
Cereals	259	Field crops	139
Fallow	74	Vegetables and potatoes	17
Legumes	8	Olive and vineyard	6
Tuber	1	Fruit trees	0.4
Forage	1	Other	9
Industrial	11		
Vegetables	9		
Fruit trees	1		
Olive	180		
Vineyard	66		
Pastures and meadows	518		
Total	1,061	Total	172

Consumption water footprint is around 9,000 hm^3 per year (figure 1). The mean footprint per person and per year (circa 1,600 m^3) is higher to that cited in the literature for the World (WWF, 2008), but is lower than Spanish average. The 80% is net imported virtual water and food and agricultural products account for a very high proportion of this footprint. Almost the 60% of consumption water footprint is due to meat consumption and production. This analysis shows that consumption in Madrid region depends from external resources; being the latter 16 times higher than the own blue water resources of the region.

Table 4: Production water footprint of Madrid region (hm^3).

Green water		Blue water		Grey water		Footprint
Irrigated crops	54	Agriculture	172	Agriculture	87	
Rainfed crops	1,007	Livestock	2	Wastewater treatment	551	
		Industry	49	Deposition	82	
		Other	82			
Total	1,061	Total	304	Total	720	2,085

This framework should be a guide for decision making about water resources management. In order to reduce the inner blue water consumption the target sector should be house holds and municipalities. Water used in agriculture represents a relative low proportion of total blue water resources and the virtual water export in grown products is low, although the total virtual water export is high because Madrid is an important logistic centre in Spain and a lot of products go through this region. Within this framework the self sufficiency is not possible because there is a high dependence on the external resources that are 16 times higher than the own blue water resources of the region. However, a more sustainable consumption should be achieved considering the consumption needs of a high populated area and the food consumption pattern, as 60% of consumption water footprint is due to meat consumption and production.

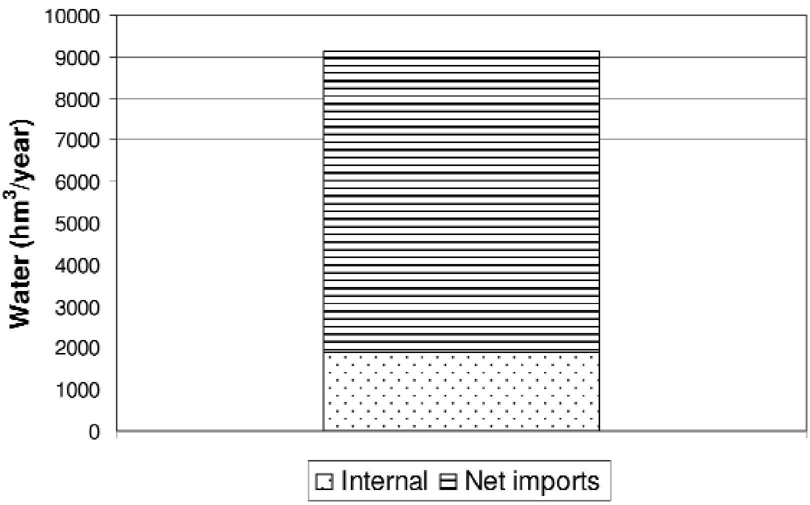

Figure 1: Consumption water footprint in Madrid region.

4. Conclusions

Production water footprint in Madrid region is approximately 2,000 hm^3 per year, almost four times the blue water available resources in the region. It is remarkable the green and grey water use. In this grey component, wastewater is the most important factor. Consumption water footprint is around 9,000 hm^3 per year from which 80% is net imported virtual water, so Madrid depends on external water resources in order to supply the actual consumption needs. These should be more sustainable reconsidering the consumption pattern, as 60% of the water footprint is due to meat production. So, this framework should be a guide for decision making about water resources management in the future.

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Carbon footprint of school meals

Namy Espinoza-Orias*, Adisa Azapagic

School of Chemical Engineering and Analytical Science
The University of Manchester, Oxford Road, Manchester, M13 9PL, United Kingdom

ABSTRACT

In the UK, around 40% of pupils eat meals served at school canteens. The Government introduced a set of standards obliging schools to implement efficient and sustainable catering systems, capable of satisfying the dietary and nutritional needs of pupils. This paper uses these systems as a basis to compare the life cycle greenhouse gas emissions of school meals and to find out how the nutritional needs of pupils could be met at a minimum impact on climate change. The carbon footprint of a 'typical' school menu prepared at two scales was evaluated and compared: in-house (small-scale) and a centralised catering facility (large-scale). The results show that the difference in the global warming impact between the two scales of meal preparation is small and that optimising school menus can reduce the impact on climate change much more significantly. Therefore, these results can help schools to choose low-carbon and nutritious menus.

Keywords: Carbon footprint, GHG emissions, school meals, catering.

1. Introduction

A fundamental service provided by governments following welfare policies is the provision of school meals to students attending publicly-funded schools. In the UK, the way this service has been provided since its introduction in 1906 by the Education (Provision of Meals) Act reflects the changes that public services have been subjected to over the years, especially with the introduction of de-regulation, privatization and the re-introduction of markets to the provision of public services (Davies, 2005). School meals, however, are but one aspect of food culture children are immersed in. Children require nutritious, healthy, safe food while growing up and at the same time learn good eating habits, hygiene and nutrition aspects and cooking skills for life.

Two key aspects illustrate the scale of the school meals provision service in the UK: the magnitude of public funds expenditure –estimated at £ 1.2 billion annually (School Food Trust, 2008) – as well as the proportion of UK school-age population being served at state-funded, maintained primary and special schools with school meals provided either by the school or a local authority: 39.3% of students in primary and special schools and 35.1% of students in secondary schools, academies and city technology colleges (Nelson *et al.*, 2009) out of a population of 7.3 million registered students in 2009 (DSCF and NS, 2009).

The UK Government established in 2005 the School Food Trust with the purpose of transforming school food and food skills. A set of nutrient-based and food-based standards specific for school meals ensued in 2006. By 2009, all food served in primary and secondary schools should be compliant whit these standards. This obligation requires the implementation of an efficient and sustainable school catering system, capable of satisfying the diet and nutritional needs of the school-age population (ages 4-18).

Current provisions for the procurement of public funded services require the consideration of the associated sustainability issues. School meals need not only to be provided in an efficient way and affordable way, but also take into consideration how social and environ-

* Corresponding Author. e-mail: Namy.Espinoza-orias@manchester.ac.uk

mental aspects may interact when decisions are taken in relation to menu planning, dietary requirements, food safety, nutritional content, training of catering staff, sourcing and seasonality of ingredients, packaging, preparation and distribution of meals, and waste management.

Two aspects affect the environmental impacts of the operation of food supply chain and food delivery systems: ingredient sourcing and mode of food preparation. The ingredients can be sourced locally, regionally or be imported; likewise, the ingredients can be produced using conventional or organic agricultural practices. As for the preparation modes, the kitchen infrastructure available in the schools determines if meals can be prepared freshly in each school (decentralised model) or if they need to be prepared at a central catering facility, transported to the schools and finalised in situ ('hub-and-spoke' model).

The present work reports the life cycle greenhouse gas emissions (GHG) associated with the two school catering systems and calculated in accordance to PAS 2050:2008 (BSI, 2008). The results can be implemented in protocols for the selection of appropriate systems for school meals provision.

2. Goal and scope of the study

The objectives of the present study were:

a) to evaluate the greenhouse gas emissions (GHG) arising from two modes of provision of school meals: in-house catering and contract catering;
b) to identify the stages within the assessed food delivery systems that contribute the most to GHG emissions; and
c) to assist decision makers at school boards, local authorities and catering contractors in the selection of the most appropriate school meals delivery system by informing them about the associated GHG emissions.

3. The functional unit and systems studied

3.1 Unit of analysis

The functional unit chosen in this study was a "a week's menu of freshly prepared meals, catering for school-age pupils and consisting of a main dish, side-dish and dessert ready to be served to a pupil at the school dining facilities".

3.2 Description of the catering systems for the provision of school meals

In the UK, school meals are provided by means of one of two major categories of catering systems: in-house catering or contract catering. In-house catering reflects the traditional approach to catering, for it is performed at a small to medium scale; it is capable of both preparing and serving food in a single operation and with little delay between these stages. In contract catering, on the other hand, food preparation is decoupled from food service. A "time buffer" is incorporated into the flow of food between the preparation and serving stages, whereby food is preserved safely and conveniently, usually by chilling or freezing. Contract catering has also adopted several methods of food processing technology such as large-scale equipment, centralized production, consistent heating and chilling treatment as well as more sophisticated packaging. Food is later regenerated at satellite kitchens, closer to the point of service (Creed, 1989; Smith and West, 2003).

3.3 Description of the scenarios analysed

In this study, two baseline scenarios representing alternative school meals catering systems and capable of delivering the same functional unit were chosen for analysis:

a) Scenario 1: In-house catering system. This scenario takes place in a medium-sized school with a student body of 350 pupils and cooking and dining facilities operated by the school. The percentage of pupils with vegetarian diet requirements is 3%. The school takes advantage of the proximity to local producers of good quality meat and vegetables and is capable of preparing daily fresh meals.

b) Scenario 2: 'Hub-and-spoke' catering system. In this scenario, the meals served to pupils from ten local schools with insufficient or non-existing kitchen facilities are prepared in a centralised catering facility and distributed by road transport on a daily basis. The scale of this catering system implies sourcing of large quantities of ingredients produced both domestically and abroad. The preparation of meals is spread over the period of a week, and some dishes/side-dishes are stored under refrigerated conditions until the day they are delivered to the schools.

Table 1 describes further the characteristics of each scenario. A sample of meals served in UK schools and complying with the nutritional guidelines for school-age children was drawn in order to construct a one-week menu rotation for analysis. Every day, pupils are served a main dish accompanied by a side-dish and followed by a dessert. Table 2 details the one-week menu rotation prepared in both scenarios and catering for omnivorous and vegetarian diets. It was assumed that all students attending the schools consumed the school meals prepared by either scenario; i.e. the take-up of meals was 100%.

Table 1. Characteristics of the scenarios under analysis

Characteristic	Scenario "Hub and spoke"	Scenario "In-house catering"
Number of meals prepared daily	3485	350
Number of students served	School size range: 302-396 students	350
Number of schools serviced	10	1
Origin of ingredients	Domestic and imported	Local
Agricultural practices used to produce ingredients	Conventional (meats, cereals, vegetables, fruits, dairy products)	Organic (meats, dairy and cereals) Conventional (fruit, vegetables)
Distance meals are transported	3 - 10 km	0 km
Percentage of vegetarian dishes	3%	3%

Table 2. Components of one-week menu rotation for analysis

Day	Meal components		
	Main dish and side dish		Dessert
	Omnivorous diet	Vegetarian diet	
Monday	Chicken curry ragout, potato wedges	Cheese & onion flan, potato wedges	Carrot cake
Tuesday	Chili con carne, rice	Cheese pasta, garlic bread	Chocolate oatcake
Wednesday	Lamb biryani, peas and sweetcorn	Vegetable cakes, peas and sweetcorn	Flapjack
Thursday	Lemon chicken risotto, salad	Homemade pizza, salad	Fruit salad and yogurt
Friday	Tuna pasta bake, garlic bread	Samosa pie, rice	Toffee apple crumble

3.4 System boundaries

The system boundaries considered for both scenarios under analysis are represented in Figure 1. The study follows a "cradle to kitchen gate" approach, with the following life cycle stages included:

a) Agricultural production of ingredients: Cultivation of arable crops, vegetables and fruit; rearing of animals (beef cattle, dairy cattle, poultry, laying hens, pigs and lambs); slaughtering of animals and processing of meat; fishing and processing of fish; primary processing of milk and dairy products; intermediate processing of ingredients.

b) Production of packaging materials: Production of cardboard, polyethylene film, polypropylene film, injection moulded plastic trays and crates and steel cans used as packaging of the ingredients.

c) Transport of raw materials: Transport of ingredients from suppliers to the catering facilities by road and sea under ambient and refrigerated conditions.

d) Preparation of meals: Preparation of ingredients, cooking, re-constitution of food, and assembly of meal components. The refrigerated and ambient storage of raw and cooked food is also included.

e) Waste management: Disposal by landfilling of waste packaging and food waste generated during the preparation of meals.

f) Delivery of meals to schools: Delivery of meals from the centralized catering facility to the schools contracting this service by road transport under refrigerated conditions.

The stages of food service, food consumption at school dining facilities, cleaning of dishware and disposal of food waste from consumption are not considered within the system boundaries.

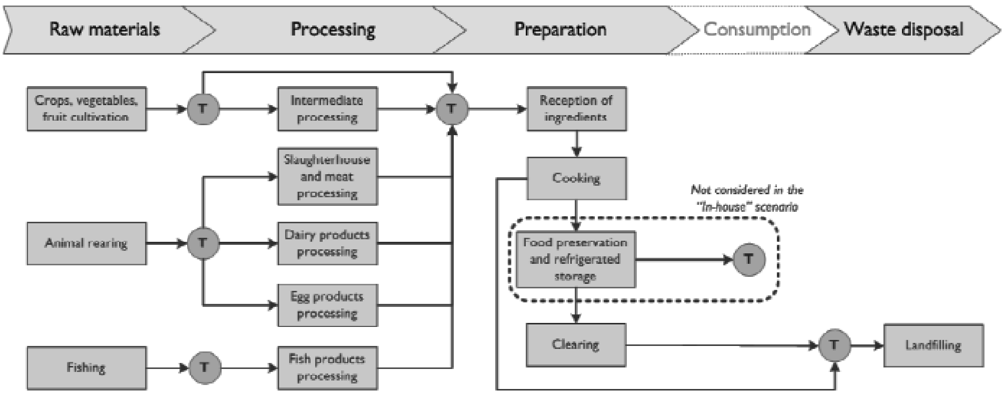

Figure 1. System boundaries (T = transport).

4. Results

Figure 2 shows the GHG emissions arising from the daily and weekly provision of school meals to students in the schools considered in both scenarios, in-house catering and contract catering. Figure 3 shows the relative contribution of each life cycle stage to the overall GHG emissions.

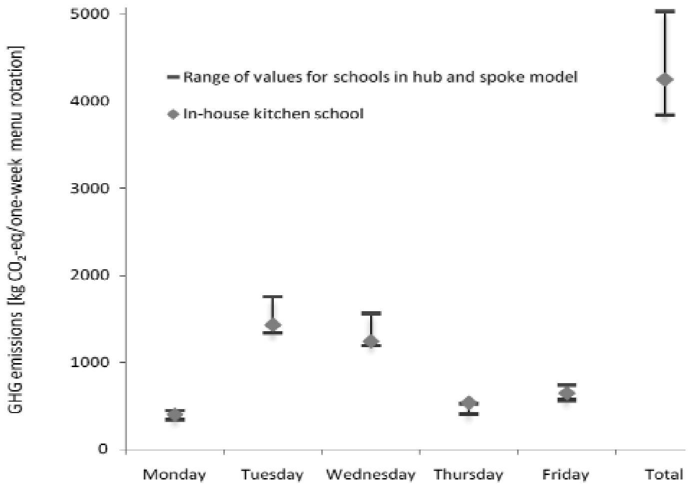

Figure 2. GHG emissions arising from the provision of meals for the schools considered in both scenarios.

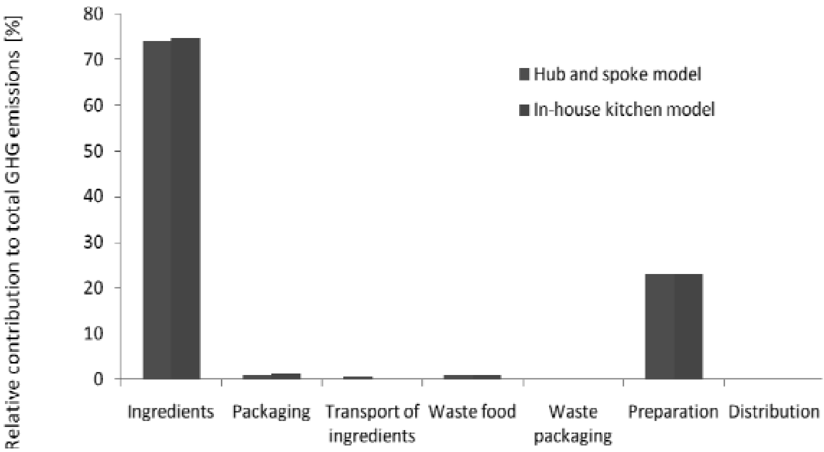

Figure 3. Relative contributions of each life cycle stage to the overall GHG emissions from the provision of school meals for both scenarios.

From Figure 2, it can be seen that the GHG emissions arising from the preparation of meals either in-house or in the centralised catering facility fall into a relatively narrow range. Therefore, it is not possible to choose conclusively one catering system over the other, given that the hub-and-spoke system serves schools located within varying distances from the centralised catering facility and with varying number of students. Clearly, though, the choice of menu recipes and daily menu planning has an effect on the total amount of GHG emissions. Although not immediately evident, factors such as budgeting, staff training, bulk buying and ordering of meals need to be incorporated into the scenarios analysed, as they also affect the GHG emissions

Two life cycle stages are shown (see Figure 3) to contribute the most to the GHG emissions of school meals provision: production of ingredients (~75%) and the preparation of meals (~23%). The variety of ingredients and the amounts required are directly related to the proportion of meat-based and vegetarian recipes included in the menu rotation. Meals can be prepared freshly on a daily basis at the school kitchen given the reduced volume of meals, thereby eliminating the need to hold food and preserve it before serving. Intuitively, it would be expected that the economies of scale for the preparation of meals would have a larger effect on greenhouse gas emissions and that the large-scale (the hub and spoke model) would have lower impacts compared to the operation of an in-house school kitchen. However, the introduction of additional energy consuming processes (food preservation and refrigerated food storage) affects the value of GHG emissions per meal. Distribution of meals to schools contributes the least, demonstrating that "food miles" are not a significant issue in the life cycle of meals prepared by the hub-and-spoke system.

5. Conclusions

The carbon footprints of school meals (at the kitchen gate) from the in-house catering and contract catering systems are on average 4412.1 kg CO_2-eq, ranging between 3837.4 and 5032.3 kg CO_2-eq per one-week menu served to schools with a student population from 300 to 400 students. The stages that contribute the most to the GHG emissions of school meals provision are the production of ingredients and the preparation of meals.

In order to understand better the differences between the catering systems assessed in this study and to explore options for the reduction of GHG emissions contributing to the carbon footprint of school meals, further work should include:

- more menu rotations including seasonal ingredients;
- reduction of meat-based dishes in the menu rotation;
- allocation of energy and water consumption per meal and food preparation process;
- allowance for increased take up of school meals (currently around 40%).

The school meals supply chain is one but a specific variant within the food supply chain, and as such, faces the challenge to reduce its contribution to the overall environmental impacts and in particular greenhouse gas (GHG) emissions. Therefore, the results obtained in this work can help the relevant stakeholders (students, parents, school directors, local authorities) to choose low-carbon menus while still satisfying the nutritional needs of pupils.

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An LCA approach for evaluating the Global Warming Potential of various food types imported to Singapore

Khoo H.H.[1,*], Wong J.L.[2], Tan R.B.H[2]

[1]Institute of Chemical and Engineering Sciences, 1 Pesek Road, Jurong Island, Singapore 627833
[2] Dept. of Chemical & Biomolecular Engineering, National University of Singapore, Singapore 117576

ABSTRACT

An LCA approach is employed to project the global warming potential (GWP) of beef, chicken, tofu, rice and tomatoes delivered to Singapore. The system boundaries covered land use, feed and energy inputs for animal farming, slaughter, cutting and freezing of meat; and agricultural land and fertilizer input for soy production and soy to tofu processing. For rice and tomatoes, the LCA process starts with agriculture, fertilizer and energy inputs for cultivation, harvesting, milling and drying (for rice), packaging, cooling (for tomatoes) and finally, transportation. The GWP results were: over 2000 kg CO_2-eq (beef); 905 kg CO_2-eq (chicken); 522 kg CO_2-eq (tofu); 219 kg CO_2-eq (rice) and 275 kg CO_2-eq (tomatoes). Based on the estimated protein intake of 50 g/day for Singapore's population of 4,987,600, the total projected reduction in GWP is estimated to be 15 million tons CO_2-eq if everyone consumes a plant-based diet once a week in a year.

Keywords: Global warming potential (GWP), agriculture, livestock, land use, food

1. Introduction

There is a growing number of research work that suggest food from livestock contributes significantly to global warming (McAlpine *et al.*, 2009). With increasing concerns on the carbon footprint of food, different types of environmental assessment tools have been applied for the purpose of projecting the environmental impact of various livestock production systems at farm level (Roy *et al.*, 2008). Singapore lacks land territories for agricultural use and imports most of its food from various countries. In this paper, LCA is used to project the carbon footprint of the following food products delivered to Singapore: i) beef from Brazil; ii) chicken from Denmark; iii) tofu from a Scandinavian Soya company in Denmark; iv) milled rice from Thailand, and v) tomatoes from Japan.

2. Methods

The functional unit selected is **1 kg protein per food**. The total amount of protein from each food type is compiled in Table 1. In the first two food life cycle production systems, the main activities taken into account are greenhouse gases from land use, energy and associated emissions for fodder/feed production, emissions throughout the lifetime of the animals, slaughter, cutting, freezing and transportation. The third LCA takes into account land use and fertilizer input, soy production, soy-to-tofu processing, freezing and transportation. Less process steps are involved in the last two food chains – the LCA system considered for rice agricultue, drying, refining and storing are from Kasmaprapruet *et al.* (2009); and LCA of tomatoes are from Roy *et al.* (2008). Missing data for tomatoe packaging and cooling are

[*] Corresponding Author. e-mail: khoo_hsien_hui@ices.a-star.edu.sg

Proceedings of LCA food 2010 (Vol. 1) • pp. 217-222

from Blengini and Busto (2009). Various scenarios of land use are not covered in all cases. Land area requirements for food production (m^2/kg food), and their associated greenhouse gas emissions, are extracted from Jungbluth *et al.* (2007) and Dalgaard *et al.* (2008). The same value of emissions due to land use (12 kg CO_2-eq/m^2) is applied throughout the paper.

The CO_2-eq characterization factors are from IPCC *Climate Change 2007: Fourth Assessment Report (AR4).* Other emissions, such as acidic gases and ammonia are not taken into account. Resource use such as water is also not considered. Emissions from ocean freight travel are taken from Gabi Life cycle engineering database.

Table 1: Protein from each food source

Food type	kg protein/kg food	Food type	kg protein/kg food
Beef/Chicken[a]	0.247	Tofu[b]	0.08
Rice[b]	0.100	Tomatoes[c]	0.0116

[a]Collins (2007); [b]USDA National Nutrient Database (2009); [c]HighProteinsFood (2004)

2.1. Beef and Chicken production

The LCA system boundary (Figure 1a) for beef starts from the farming stages (land use), production of feed, raising of calves to maturity, slaughtering and cutting, grinding and freezing, and finally, shipment to Singapore. The life cycle stages and main input-output data for beef production are extracted from Cederberg *et al.* (2009). Each calf takes up to 4 years to reach the age for slaughtering. The average carcass weight per cow is 184 kg. The energy required for freezing is taken from Raminez *et al.* (2006). Land used for grazing is estimated as 33.0 m^2/kg beef with 12 CO_2-eq/m^2 emissions from land use (Jungbluth *et al.*, 2007).

The next case study is on chicken production from Denmark (Halberg and Nielsen, 2003; Pontoppidan and Hansen, 2000). The life cycle stages include land use for farming, production of chicken feed, raising broiler, slaughtering and cutting, storage and finally transportation from Denmark to Singapore (Figure 1b). Land use is estimated to be 12.5 m^2/kg-chicken. For chicken feed, land use is estimated as 3.93 m^2/kg soy and 2.07 m2/kg wheat (Dalgaard *et al.*, 2008).

The compiled input-output data for both beef and chicken are shown in Table 2.

2.2. Tofu, rice, tomatoes

Tofu, or bean curd, is one of the many products processed from soy beans. It is a common sight in Asian dishes. The processing of tofu from soy is rather complex and demands considerable energy inputs (Håkansson *et al.*, 2005). The life cycle stages starts with farming and soy production, with land use of 3.93 m^2/kg soy. The use of fertilizer, which adds to N_2O emissions, is required (Dalgaard *et al.*, 2008). The next LCA stages are the processing of soy-to-tofu. About 0.56 kg-soy beans are required to make 1 kg tofu. More details of the production stages can be found from Advameg (2009) and EcoInvent (2009). A summary of the flow diagram for the production of tofu from soy is shown in Figure 2.

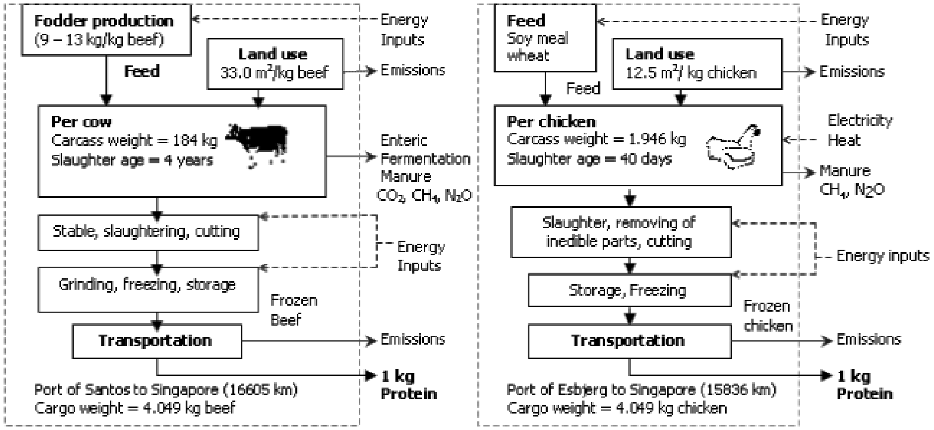

Figure 1. Flowchart of: (a) beef production, and (b) chicken production

Table 2: Compiled input-output data for beef and chicken production

Beef (Input)	**Beef (output)**	**Chicken (input)**	**Chicken (output)**
Feed: 11 kg fodder /kg beef	Enteric Fermentation: 250 g CH_4 /kg beef	0.514 kg soy meal + 1.44 kg wheat/kg chicken	Manure CH_4: 5.66×10^{-4} kg /kg chicken
Fodder production: 46.8 MJ/kg beef	Manure: 4.62 g CH_4 /kg beef	Feed production: 2.2 MJ/kg chicken	
Slaughtering, cutting: 10.8 MJ/kg beef	Direct emissions: 5.71 g N_2O/kg beef	Electricity/Heat: 1.91 MJ/kg chicken	Manure N_2O: (indirect) 1.25×10^{-5} kg /kg chicken
Storage/Freezing: 14.4 MJ/kg beef	Indirect emissions: 0.20 g N_2O/kg beef	Slaughter, cutting, storage, freezing: 2.17 MJ/kg chicken	
	Frozen beef		**Frozen chicken**

Rice production, from Thailand (Kasmaprapruet *et al.*, 2009), starts with seeding, cultivation and finally, harvesting. Fertilizers (ammonium sulphate) are used. Next process steps include drying, milling, and finally storage (Figure 3a). Grain yield is 6.12 t/ha. Greenhouse gas emissions from paddy fields are reported by Blengini and Busto (2009) as 25.7 – 107.1 g CH_4 per kg rice and 0.2 g N_2O per kg rice. Tomatoes from Japan are produced by the local farmers and shipped to Singapore. The LCA steps and data, from Roy *et al.* (2008), are simplified in Figure 3b. Fertilizer and energy inputs are also used. The input-output data for tofu-from-soy, rice and tomatoes are shown in Table 3.

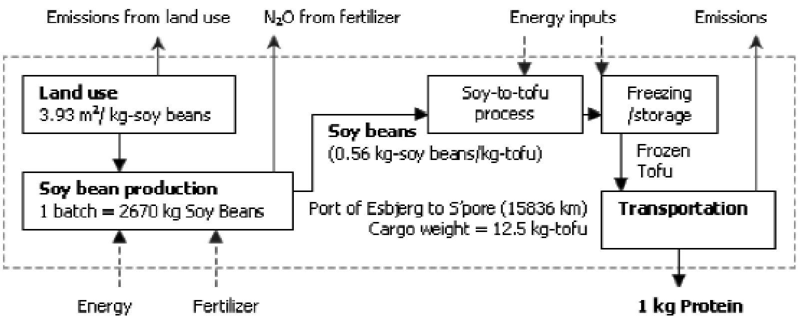

Figure 2: Flowchart of soy-to-tofu production

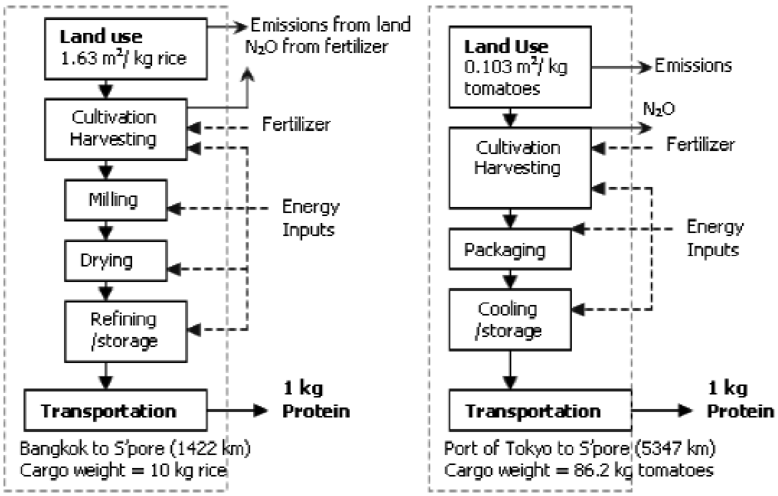

Figure 3. Flowchart of (a) Rice production, and (b) Tomato production

Table 3: Compiled input-output data for tofu from soy, rice and tomatoes

Soybean production (1 batch = 2670 kg soy beans)		**Tofu from Soy**	
Input	**Output**	**Input**	**Output**
Traction: 1491 MJ/batch	1 batch soy beans	0.56 kg soy beans	**1 kg tofu (frozen)**
Fertilizer: 230 kg N/ha	N_2O: 4.7 kg/batch	Process energy: 43.2 MJ/kg-tofu	
Rice		**Tomato**	
Input	**Output**	**Input**	**Output**
Fertilizer: 0.000255 kg/kg rice	Field emissions: 0.0002 kg N_2O/kg rice 0.0664 kg CH_4/kg rice	Agriculture activities: 0.42 MJ diesel/kg	Emissions: 0.0034 kg N_2O/kg
Harvest, Milling: 0.25 MJ /kg 0.25 MJ/kg	**Milled white rice (packed)**	Fertilizer: 0.197 kg/kg	Other emissions (from Packaging/Cooling): 0.06 kg CO_2/kg
Drying: 2.4 MJ /kg 6.67×10^{-4} MWh/kg			**Tomatoes (packed)**
Refining, Packaging: 0.278 MJ /kg 7.69×10^{-5} MWh/kg			

3. Results and discussions

The results of CO_2-eq emissions for all five food options (**per kg protein**) are presented in Figure 4. As expected, the largest global warming results are from beef (over 2000 kg CO_2-eq), followed by chicken (approx. 900 kg CO_2-eq) per protein. Another study by Weber and Matthews (2008) on food supply chain confirmed that the contributions to climate change were dominated by red meat, and very little from transportation. Next highest is tofu and tomatoes (approximately 500 kg CO_2-eq and 300 kg CO_2-eq respectively). The only apparent transportation emissions are from tomatoes. This is because a large amount of tomatoes (82.6 kg) are required to provide the equivalence of 1 kg protein.

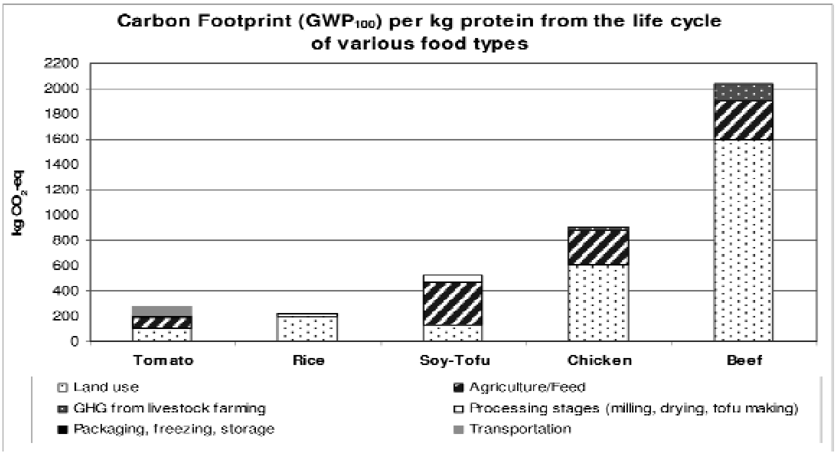

Figure 4: Carbon Footprint Results for various Food Production

One of the significant portions of the graphs are from land use, especially for beef production. Globally, cattle grazing have been reported to be a major driver for deforestation; one of the biggest environmental damage related to livestock farming is found to be the large scale clearing of forest area in the Amazon (McAlpine *et al.*, 2009). It should be highlight that in this study, different land use scenarios are not covered and a moderate estimate of 12 CO_2-eq/m^2 emitted due to clearing of forest area is applied for *all five cases* (Jungbluth *et al.*, 2007). For tofu, the largest contribution to global warming is fertilizer use in agriculture. To further reduce GWP amounts from mineral fertilizers used in agriculture, organic farming for soya beans production is suggested. This alternative not only prevents N_2O emissions from fertilizer use, it further reduces CO_2 emissions by soil sequestration (Müller and Davis, 2009). In addition, organic agriculture has also proven to have increased yield per m^2 land (Badgley *et al.*, 2007), which is an important factor in meeting growing food demands globally in a sustainable manner.

Based on the estimated protein intake of 50 g/day for Singapore's population of 4,987,600 in 2009 (Statistics Singapore, 2009), the total projected reduction in greenhouse gas emissions is estimated to be 15 million tons CO_2-eq if everyone consumes a plant-based diet once a week in a year.

4. Conclusions

The global warming results display a large amount of greenhouse gas reductions from beef to chicken, and chicken to tofu, rice and tomatoes. An extremely simple and straightforward adoption of a once-a-week meatless diet could have huge carbon savings. If organic agriculture is introduced for plant-based food, the carbon savings are expected to be even more tremendous. Apart from its significant contribution to global warming impacts, the livestock industry has long been creating large scale environmental damages associated with the loss of biodiversity, degradation of land, and loss and pollution of water resources. Further studies should focus on the GWP of other protein sources (e.g., organic produce, pork, or seafood).

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Estimating the carbon footprint of the Galician fishing sector (NW Spain)

Diego Iribarren, Ian Vázquez-Rowe, Almudena Hospido*, María Teresa Moreira, Gumersindo Feijoo

Dept. of Chemical Engineering, University of Santiago de Compostela, Spain

ABSTRACT

The food production system accounts for a relevant portion of the greenhouse gas emissions associated with any country. In this context, there is an increasing market demand for information regarding the global warming impact of consumer food products. This study deals with the carbon footprint assessment of the fishing sector as a key subgroup in the food industry. The analysis is based on a representative set of species within the Galician fishing sector (NW Spain), including species obtained from coastal fishing (horse mackerel, Atlantic mackerel, European pilchard and blue whiting), offshore fishing (European hake, megrim and anglerfish), deep-sea fishing (tuna), extensive aquaculture (mussels) and intensive aquaculture (turbot). The carbon footprint associated with each species was quantified by following a business-to-business approach on the basis of 1 year of fishing activity. These individual carbon footprints were used to estimate the carbon footprint of the Galician fishing sector.

Keywords: aquaculture, carbon footprint, fishery, global warming, seafood.

1. Introduction

The food production system as a whole is recognized as one of the major contributors to environmental impacts (Foster *et al.*, 2006). Food production, processing, transport and consumption account for a relevant portion of the environmental greenhouse gas (GHG) emissions associated with any country (Garnett, 2008). The present paper deals with the assessment of the carbon footprint of the fishing sector as a key subgroup of the food industry. In particular, the estimation of a value for the global warming impact of the Galician fishing sector is developed (Iribarren *et al.*, 2010a).

The main Spanish region regarding fishing is Galicia (NW). Galician fishing activities constitute a key economic sector that provides 10% of the regional GDP. This sector usually distinguishes two main activities: commercial fishing (that comprises the coastal, offshore and deep-sea fisheries) and aquaculture (that encompasses extensive and marine intensive farming practices) (Xunta de Galicia, 2009).

Life Cycle Assessment (LCA) is a suitable methodology to undertake the environmental assessment of seafood products (Pelletier *et al.*, 2007), being the global warming impact category among the most common impact categories assessed. However, current trends in the communication of the climate change indicator have led to the development of Carbon Footprinting (CF) as an independent methodological approach.

CF estimates the overall amount of GHG emissions associated with a product along its supply chain (Carbon Trust *et al.*, 2008). Among the standardized methods developed, the Publicly Available Specification 2050:2008 is highlighted as it is receiving increasing acceptance (BSI, 2008). It specifies requirements for the assessment of the life cycle GHG emissions of goods and services based on key life cycle techniques and principles (Sinden, 2009).

* Corresponding author e-mail: almudena.hospido@usc.es

2. Case study

Regarding the Galician fishing sector, commercial fishing covers 60% of the regional fishing production rate and 75% of the economic turnover (Xunta de Galicia, 2009). Moreover, the role played by aquaculture should not be disregarded as it provides 40% of the total production and 25% of the total economic turnover.

The current case study performs the CF of selected species targeted by the Galician fishing sector. This selection was decided so that representativeness was guaranteed (92% of the total production as well as 79% of its economic turnover), then enabling the subsequent estimation of a global carbon footprint for the whole Galician fishing sector following a bottom-up approach. According to the current distribution of this sector, a representative set of species was defined:

(i) Coastal fishing: horse mackerel (trawling and purse seining), Atlantic mackerel (trawling and purse seining), blue whiting (trawling), hake (trawling), European pilchard (purse seining) and chub mackerel (purse seining).

(ii) Offshore fishing: hake (long lining and trawling in the Northern Stock); megrim, anglerfish and Norway lobster (trawling in the Northern Stock); conger eel, Atlantic pomfret, common ling, rock fish, fork beard and splendid alfonsino (long lining in the Northern Stock); porbeagle, mako shark, bigeye tuna, blue shark and swordfish (long lining in the Azores).

(iii) Deep-sea fishing: skipjack and yellowfin tuna (purse seining in the Indian, Atlantic and Pacific Oceans).

(iv) Extensive aquaculture: mussels cultured in traditional rafts.

(v) Marine intensive aquaculture: turbot.

3. Materials and methods

Following a business-to-business (B2B) approach (Carbon Trust *et al.*, 2008), the GHG emissions from capture/culture to landing in Galician ports were included for the CF of commercial fishing and extensive aquaculture species. Regarding marine intensive aquaculture, the GHG emissions arising from the production of farmed turbot were assessed up to the supply of commercial adult turbot for transport to retailers. In all cases, the carbon footprint was referred to 1 year of activity.

Inventory data for the assessed coastal species were obtained through a series of questionnaires filled out by skippers from a wide range of Galician fleets (Vázquez-Rowe *et al.*, 2010a). The assessed fleets included coastal purse seiners (30) and coastal trawlers (24). Similarly, data regarding the offshore species were obtained through an analogous questionnaire (Vázquez-Rowe *et al.*, 2010b). In this case, the interviewed skippers belonged to the Northern Stock fleet (12 long liners and 9 trawlers) and to the Azores long lining fleet (5). On the other hand, inventory data for Galician deep-sea fishing were obtained from the tuna purse seiners (9) that land in Galician ports (Hospido and Tyedmers, 2005).

Main data for mussel culture in traditional rafts in Galicia were obtained from 22 auxiliary vessels in charge of 80 rafts (Iribarren *et al.*, 2010b,c), while inventory data for marine intensive aquaculture were mainly taken from the environmental statements of several Galician plants (Iribarren *et al.*, 2010d).

Secondary data came from the ecoinvent database as it is considered the most complete and updated (Frischknecht *et al.*, 2007).

Following the PAS 2050 requirements (BSI, 2008), capital goods were excluded, economic allocation was applied (when necessary) and the values of global warming potentials

(GWP100) to transform GHG emissions in kg of CO_2e were in accordance with the latest ones available from the Intergovernmental Panel on Climate Change (IPCC, 2007).

4. Results

4.1. Commercial fishing

Table 1 gathers the carbon footprints for the coastal fishing species caught in Galicia. The first eight rows involve the evaluated species, whereas the ninth row corresponds to a rough estimate for the non-evaluated species based on the fishing gears used for the extraction. Similarly, Table 2 presents the carbon footprints related to offshore fishing. The first 16 rows refer to the evaluated species, while the next row involves an estimate for the non-evaluated species according to the fishing gear and the fishing area.

Table 1: Carbon footprint calculation for the annual Galician coastal fishing

Species	Fishing gear	Catch rate (t/y)	Carbon footprint (t CO_2e/y)
European pilchard	Seining	15,022	11,071
Atlantic horse mackerel	Trawling	12,898	11,350
Atlantic horse mackerel	Seining	11,246	10,447
Atlantic mackerel	Trawling	9,795	5,358
Atlantic mackerel	Seining	6,284	3,632
Hake	Trawling	11,094	44,267
Blue whiting	Trawling	12,838	12,273
Chub mackerel	Seining	8,811	6,511
Non-evaluated species	Varied	23,648	32,849
TOTAL	-	111,636	*137,759*

Table 2: Carbon footprint calculation for the annual offshore fishing in Galicia

Species	Fishing gear	Fishing area	Catch rate (t/y)	Carbon footprint (t CO_2e/y)
Hake	Lining	Northern Stock	9,770	55,689
Hake	Trawling	Northern Stock	5,555	34,775
Megrim	Trawling	Northern Stock	6,437	48,730
Anglerfish	Trawling	Northern Stock	4,282	40,162
Norway lobster	Trawling	Northern Stock	695	17,711
Conger eel	Lining	Northern Stock	2,050	6,293
Atlantic pomfret	Lining	Northern Stock	3,583	9,890
Common ling	Lining	Northern Stock	778	1,922
Rock fish	Lining	Northern Stock	1,182	6,487
Fork beard	Lining	Northern Stock	1,342	6,630
Bigeye tuna	Lining	Azores	127	1,684
Splendid alfonsino	Lining	Northern Stock	80	222
Mako shark	Lining	Azores	181	1,064
Porbeagle	Lining	Azores	480	2,815
Swordfish	Lining	Azores	782	7,252
Blue shark	Lining	Azores	1,807	3,956
Non-evaluated species	Varied	Varied	9,843	31,720
TOTAL	-	-	48,973	*277,004*

Finally, Table 3 shows the carbon footprint of tuna according to the ocean where capture takes place. The sum of the three carbon footprints for tuna adds up to the value that is assumed to represent the carbon footprint of the annual Galician deep-sea fishing.

Table 3: Carbon footprint calculation for the annual deep-sea fishing activity in Galicia

Species	Fishing gear	Ocean	Catch rate (t/y)	Carbon footprint (t CO_2e/y)
Tuna	Seining	Atlantic	38,038	53,481
Tuna	Seining	Indian	70,800	85,314
Tuna	Seining	Pacific	26,068	44,341
TOTAL	-	-	134,906	*183,136*

4.2. Aquaculture

Calculating the carbon footprint of mussel production (0.083 t CO_2e/t) and taking into account that mussel production (188,818 t/y) involves near all the production of Galician extensive aquaculture (195,103 t/year), the carbon footprint estimate for this subsector is 16,213 t CO_2e per year.

In a similar way, determining the carbon footprint of farmed turbot in Galicia (19.40 t CO_2e/t) and taking into account that turbot production (6,863 t/y) practically represents the total Galician marine intensive aquaculture (7,144 t/y), the carbon footprint estimate for this subsector is 138,592 t CO_2e per year.

4.3. Galician fishing sector

The lump sum of the individual fishing and aquaculture subsectors provides the carbon footprint of the Galician fishing sector: 752,705 t CO_2e per year.

5. Discussion

CF involves a number of interesting applications and provides chain transparency and accountability for seafood (Ayer *et al.*, 2009). Thus, the value of the individual carbon footprints as well as the total carbon footprint of the Galician fishing sector is highly useful for benchmarking purposes. In this sense, a benchmark (i.e., a reference value) for measuring and communicating emission reductions is provided for the whole Galician fishing sector.

As observed in Figure 1, coastal and deep-sea fishing contribute to the global carbon footprint in an expected manner according to their catch rate and economic turnover shares for the Galician fishing sector. However, offshore fishing presents a carbon footprint contribution higher than that expected on the same basis. Finally, while extensive aquaculture entails a low contribution to the total carbon footprint, marine intensive aquaculture shows an opposite behaviour.

The use of CF for the Galician fishing sector allows the different stakeholders to prioritize opportunities to reduce the related GHG emissions. In this regard, efforts in the field of offshore fishing and marine intensive aquaculture should be primarily encouraged. Furthermore, the individual CF studies for species from commercial fishing and extensive aquaculture suggest that, as expected, diesel production and use constitutes the main source of global warming. On the other hand, when dealing with climate change mitigation for intensive aquaculture practices, improvement potentials should be centred on the minimization of the energy demand for the operation at aquaculture plants.

Despite these interesting potentials, CF is not lacking in methodological limitations. In this sense, although carbon footprints are presented as sole figures, caution is needed when reporting this type of results as assumptions and methodological choices can highly determine the final value (e.g., exclusion of capital goods and allocation procedure choices). Additionally, the use of CF involves the restriction of the environmental performance of a product to the global warming impact category so, when pursuing a comprehensive environmental assessment, the complementary use of LCA is recommended.

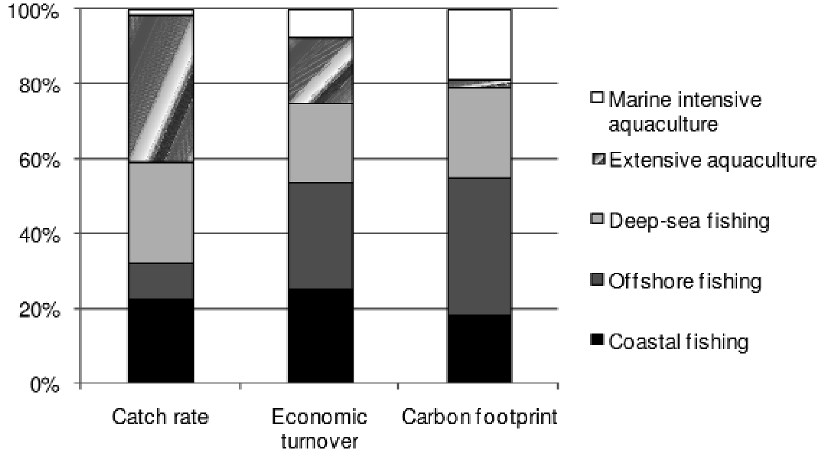

Figure 1: Contribution per subsector to the total catch rate, economic turnover and carbon footprint of the Galician fishing sector

6. Conclusions

CF has proved to be a useful tool for the assessment of the life cycle GHG emissions related to the Galician fishing sector. Through the calculation of the individual carbon footprints for a selection of species, the carbon footprint of the whole Galician fishing sector was estimated according to a bottom-up approach. An extended collection of environmental information on seafood was then provided.

The application of CF to the Galician fishing sector led to identify opportunities for climate change mitigation. Offshore fishing and marine intensive aquaculture were found to be the subsectors where improvement actions are primarily encouraged.

Being aware of CF limitations, a general strength of its wide use lies in the spread of life-cycle thinking. Thus, a responsible use of CF would stimulate the establishment of a more thorough framework for the environmental assessment of products.

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Comparing rice products: confidence intervals as a solution to avoid wrong conclusions in communicating carbon footprints

Thomas Kägi[1,*] , David Wettstein[2], Fredy Dinkel[3]

[1] myclimate, Sternenstrasse 12, 8002 Zürich, Switzerland
[2]myclimate, Sternenstrasse 12, 8002 Zürich, Switzerland
[3]Carbotech AG, 4002 Basel, Switzerland

ABSTRACT

Declaration of product carbon footprints often leaves the interpretation to the consumers, which may lead to wrong conclusions such as implying a difference where there is none. This study shows the outcome of a rice life cycle assessment (LCA) case study with and without confidence intervals of the results. Primary data of the industries combined with ecoinvent inventories for secondary data were used. Considering the appropriateness of the data being used, errors of the in– and output processes were considered according to the pedigree matrix used in ecoinvent. The simplified error analysis of the Environmental Management and Information System (EMIS) was used. Results presented without confidence interval imply that organic rice has the highest carbon footprint, whereas results with confidence interval show that there is no significant difference. Therefore, the inclusion of data quality is crucial to reduce wrong interpretation and to cultivate the acceptance of LCA.

Keywords: confidence interval, product comparison, carbon footprint, rice, uncertainty considerations

1. Introduction

Declaration of product carbon footprints is becoming more and more popular. Tesco in UK as well as Casino in France are two examples of companies who started to declare the carbon footprint of some of their products. Often, detailed numbers are given, with no more information for interpretation. Some consumer may start to compare the carbon footprints of similar products intending to choose the more ecologically or environmentally friendly product. Besides the fact that the carbon footprint does not cover the total environmental impact of a product, the comparison may lead to wrong conclusions such as implying a difference where there is none, if the data inventory quality and methodological uncertainty is considered. This rises the question how consumers can be given carbon footprint information without misleading potential.

2. Methodology

2.1 Goal and scope

The goal of this study is to show and analyse the different outcomes of a carbon footprint case study comparing rice products with and without confidence intervals of the results.

* Corresponding Author. e-mail: thomas.kaegi@myclimate.org, david.wettstein@myclimate.org

All relevant life cycle phases were considered. The cultivation of conventional and organic rice from Italy, conventional rice from USA and upland rice from Switzerland (Canton of Ticino) was assessed in this study. All rice types were dried and stored, refined and packed. For conventional rice, a scenario of parboiled rice was considered as well. Transport to Switzerland was accounted for the rice types. The cooking of the rice was included following the instruction on the package and using an average energy mix for Swiss cooking stoves. Finally, disposal of the packaging materials to incineration was taken into account.

The functional unit is 1 kg of rice as it is available in the store (processed rice in dry condition).

2.2 Inventory

Data for Italian rice cultivation was taken from Blengini & Busto (2009). Rice cultivation in the USA was taken from the ecoinvent report No 15b. (Kägi & Nemecek 2009) and data for rice cultivation in Switzerland was collected from local experts in the Magadino region in the Canton of Ticino. Methane emissions from the rice fields were estimated with data from Blengini & Busto (2009) for Italian rice and with data from Kägi & Nemecek (2007) for US rice. 2g of methane emissions per kg of rice were conservatively estimated for upland rice that is not flooded, has no anaerobic conditions in the field and cultivated similarly to wheat (Grund für diese Quantifizierung?). Other direct field emissions such as ammonia, nitrous dioxide or nitrate were calculated according to Nemecek & Kägi (2007).

The ecoinvent inventory V2.1 database (Swiss Centre for LCA 2009) was used for other secondary data (e.g. fertiliser production) and the emission factors.

2.3 Impact Assessment

The global warming potential (GWP) with a time horizon of 100 years according to IPCC 2007 was considered. The LCA was performed using the software EMIS (Environmental Management and Information System) developed by Carbotech AG (Dinkel 2009).

2.4 Uncertainty Considerations

To describe the uncertainty of data and model calculations distribution functions like normal or lognormal distribution are used. Especially for emissions where the distributions typically are not symmetric the lognormal distribution is a better approximation than the normal distribution. But the advantage of normal distributions is that there are analytic functions to calculate the error propagation over the process chain if the errors are independent of each other what is mostly the case. In contrast to the lognormal distributions where methods like Monte Carlo simulations have to be used to calculate the overall errors. So using normal distributions the results can be calculated in seconds instead of hours. This is one of the main reasons why in EMIS a simplified error calculation using normal distribution function is used. Leading to the effect that the error propagation will be calculated and the user gets always an estimation of the confidence intervals of the LCA results. Another reason is, that the error distributions have quit always to be estimated so also a lognormal distribution is not the truth and good estimation is always better than nothing and even today there are few LCA studies giving the uncertainties of the results even if there are leading software tools giving the opportunity to do an error calculation with Monte Carlo simulation.

The methodological uncertainties of the GWP according to IPCC (2007) were assumed to be zero for CO_2 and about 10% for N_2O and CH_4. Considering the appropriateness of the data being used, errors of the in– and output processes were taken into account. Errors of the in- and output processes were defined according to the pedigree matrix used in ecoinvent (Swiss Centre for LCA 2009). The error intervals are presented on the 68% level (standard deviation).

3. Results

The results show that the direct field emissions and especially methane are the main contributors to the GWP, with the exception of upland rice cultivation (Figure 1). Upland rice needs more inputs (e.g. fertilisers, diesel for agricultural machinery) per kg output but generates almost no methane emissions because the fields are not flooded and anaerobic soil conditions are avoided. The second highest impact comes from the cultivation and the parboiling process for parboiled rice. Other processes such as refining and cooking are of lesser importance. Transport emissions are to some extent relevant for imported rice from overseas. The packaging and its disposal is almost irrelevant if looking at the whole carbon footprint, although cardboard boxes (US and Swiss rice) have a lower GWP than plastic bags (Italian rice).

The results indicate that organic rice has the highest carbon footprint per kg rice mainly because of the methane emissions, followed by parboiled rice and white rice from USA and Italy. Upland rice shows by far the best performance considering the GWP.

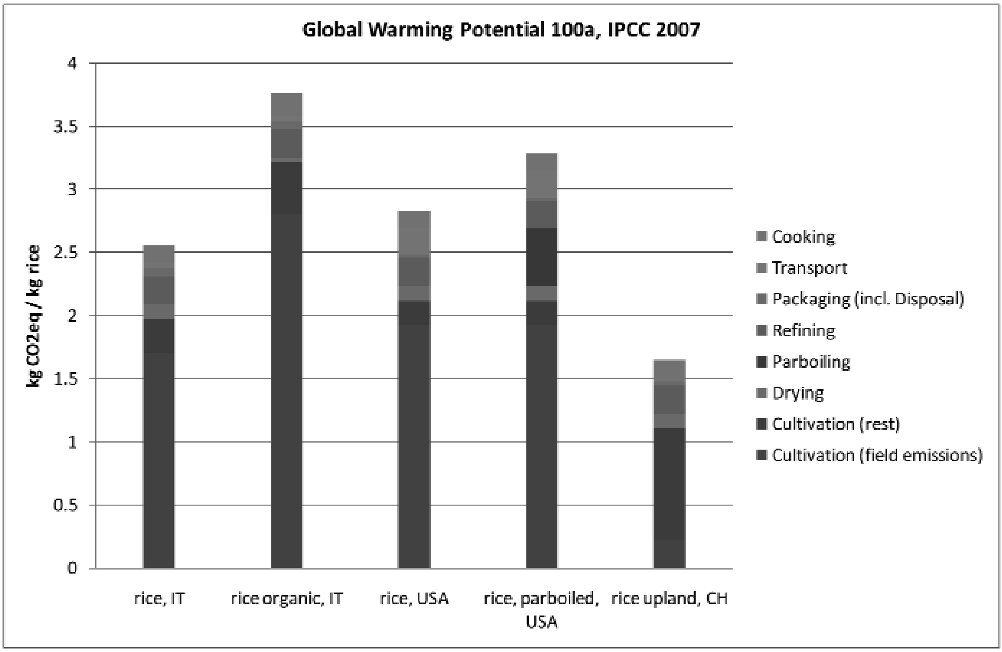

Figure 1: Global warming potential of the considered rice products and its processes without confidence interval.

4. Discussion

Presenting the results without associated confidence intervals lead to the conclusion that organic rice has the highest carbon footprint per kg of rice,. Conventional rice seems to have a better carbon footprint than organic rice even if parboiled. Furthermore, rice from the USA seems to have a higher carbon footprint than Italian rice. Upland rice which is cultivated similarly to other grain crops does not need flooded fields. Therefore, there are almost no methane emissions due to anaerobic soil conditions and the GWP per kg rice is much lower than with flood-cultivated rice, whether it is conventionally or organically cultivated.

Taking into account the confidence intervals of the results leads to other conclusions (Figure 2). There is now no significant difference between organic and conventional rice, whether it is parboiled or not. Furthermore, the results show that there is no difference between conventional rice from the USA and Italy, although transport distances vary a lot. Only upland rice still shows a significantly lower carbon footprint than all the flood-cultivated rice products.

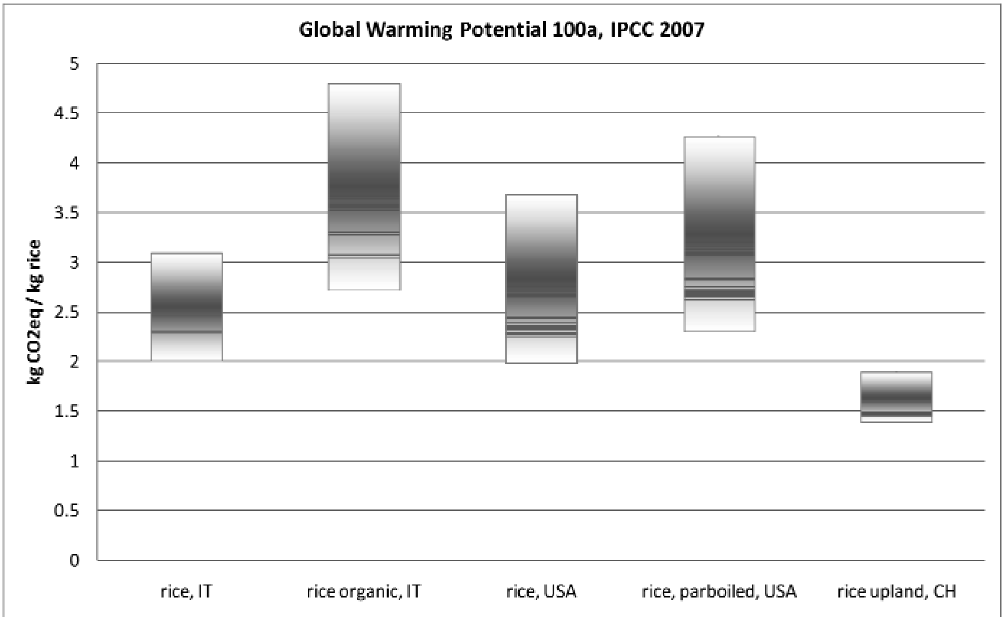

Figure 2: Global warming potential of the considered rice products with confidence interval.

Because of the sometimes high data uncertainty, especially of the direct field emissions, the confidence intervals vary from 15% (upland rice) to 31% (organic rice) for the results.

A comparison on the 68% level (standard deviation) illustrates, that upland rice has a lower GWP per kg rice than the other rice products with a probability of at least 70%. The opposite hypothesis, that upland rice has a higher carbon footprint than the other rice products, corresponds to a probability of less than 8 %.

Wrong conclusions cannot be totally avoided. Addressing data uncertainty issues when performing LCA's, and including confidence intervals into the presentation of results, may minimize the risks of wrong conclusions, however.

There is another benefit from the simplified error analysis: The error analysis can be seen as an indicator of how well the data quality fits the scope of a study. It indicates to some extend if the data gathering was precise enough for the LCA comparison. Better data quality leads to lower uncertainty of the results. For some product comparisons rough data might be enough in order to show significant differences (e.g. upland and conventional rice). For more similar products (e.g. conventional rice products), data need to be of much better quality in order to still define significant differences in the results.

5. Conclusion

This study signifies the importance of the inclusion of data quality considerations if results are communicated. This procedure is crucial to cultivate the acceptance of LCA and to reduce wrong interpretation of the results. The simplified error analysis is a helpful tool for assessing and defining very efficiently the necessary data quality considering the scope of a project. But there are some limitations such as the assumption that the data errors have a normal deviation that need further development. However, the question remains how to communicate such uncertainty results to consumers making sure they understand the message.

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Calculation of CO_2 equivalent emissions in agri-food sector applying different methodologies

Patricia Pascual[1], Carles M. Gasol[1,2], Julia Martínez-Blanco[1,2 *], Jordi Oliver-Solà[1,2], Pere Muñoz[3], Joan Rieradevall[1,2,4], Xavier Gabarrell[1,2,4]

[1] Inèdit Innovació S.L. Parc de Recerca de la UAB. Carretera de Cabrils km2 (IRTA). 08348 Cabrils (Barcelona), Spain

[2] SosteniPrA-Institut de Ciència i Tecnologia Ambientals (ICTA). Universitat Autònoma de Barcelona. Campus de la UAB. 08193 Bellaterra (Barcelona), Spain.

[3] SosteniPrA-Institut de Recerca Tecnologia Agroalimentàries (IRTA), Horticultura ambiental. Ctra. de Cabrils, km2. 08348 Cabrils (Barcelona), Spain.

[4] Departament d'Enginyeria Química. Universitat Autònoma de Barcelona. Campus de la UAB. 08193 Bellaterra (Barcelona), Spain.

ABSTRACT

Nowadays, evaluating the environmental behavior of products has become an essential issue, not only to fight against global warming and other Earth threats but also because consumers and administrations demand this information. The tools that allow us to calculate it are, among others, Life cycle assessment (LCA) and Carbon footprint (CF). Presently Carbon footprint methodology is gaining strength in the field of environmental assessment. In this paper, LCA and CF methodologies have been performed for the Mediterranean tomato production, using two cultivation options, whether greenhouse technology is used or not. The CF methodology considered was Public available specification (PAS) 2050:2008. The aim of this paper is to determine whether CF is an appropriate methodology for the analysis of agri-food systems or not.

Keywords: PAS2050, LCA; Carbon footprint, Mediterranean tomato production, Greenhouse cultivation, Eco-label.

1. Introduction

The agri-food sector is an essential activity for the survival of mankind. As a response to the growing demand of an increasing population, this sector has changed from sustainable processes to much more intensive methods. Consequently food production has become an important contributor to the depletion of natural resources and climate change (Nonhebel, 2004). Not only the production process but also food consumption involve greenhouse gases (GHG) emissions. The rise in GHG emissions has resulted in global warming, one of the most relevant environmental issues today for policy makers.

In this context, it is necessary to quantify the environmental behavior, and particularly the CO_2 eq. emissions, of products. There are many different tools for monitoring and managing GHG emissions, but a large part of scientific community considers LCA as the most appropiate approach to assess environmental impact. Another tool increasingly used is CF.

LCA is applied to calculate the potential impacts through the whole life cycle of the product (ISO, 2006). The results obtained from a LCA study include several environmental indicators such as Abiotic depletion, Global Warming Potential (GWP), Ozone Layer Depletion or Cumulative energy demand.

[*] Corresponding author: julia.martinez@uab.cat Tel: (34)935813760. Fax: (34)935868008.

On the other hand, CF methodology is confined to analyze the GHG emitted by a product during its life cycle. It is quantified using one indicator, GWP (EPLCA, 2007; Carbon trust *et al.*, 2008) and can be calculated through several methodologies. In this paper PAS 2050 has been chosen mainly because it has been previously used to evaluate a large amount and different type of products. Nowadays CF is becoming a popular tool for some reasons. First, the existence of online calculators that have sprung up the estimation of personal footprints (Johnson, 2008). Second, the results can be easily converted to an eco-label. The third reason is that CF analysis is limited to GHG emissions which make the study shorter and cheaper.

For these reasons, identifying the critical points of a product by means of LCA or CF has become an interesting discussion. In this paper, Mediterranean tomato production considering different cultivation technologies has been assessed in order to evaluate whether CF is an appropriate methodology for analysis of agri-food systems. The tomato crop has been chosen because it is a relevant product in the Mediterranean agri-food sector and has been previously studied by the research group. This paper is based on the study carried by Martínez-Blanco *et al.* (2010), in which the production process of tomato was assessed.

2. Environmental tools

In this section LCA and CF are described, with a focus on the main differences between them.

2.1. Life cycle assessment (ISO 14040)

As abovementioned, LCA involves the evaluation of a product system to determine its environmental impacts. It is based on the ISO 14040 series and divided into four steps: goal and scope, inventory analysis, impact assessment and interpretation of results (ISO, 2006).

The impact assessment was made according to CML 2001 baseline method (Guineé, 2001) which gives a list of impact categories to be studied (ISO, 2006). Taking into account the main aim of this paper, just the GWP indicator is analyzed and compared to CF.

2.2. Carbon footprint (PAS2050)

CF has been calculated following PAS 2050:2008 methodology, developed by BSI and co-sponsored by the Carbon Trust and the Department for Environment, Food and Rural Affairs of the United Kingdom. It is based on BS EN ISO 14040, BS EN ISO 14044 (ISO 2006). Two approaches can be made (Carbon Trust *et al.*, 2008): (i) Business-to-consumer: from raw materials extraction to consumer use and final disposal/recycling and (ii) Business-to-business: CF stops when the product is delivered to other manufacturer. The latter approach has been chosen in order to have the same system boundaries as the LCA carried out by Martínez-Blanco *et al.* (2010). Four steps are used to calculate the CF following PAS 2050 method (Carbon trust *et al.*, 2008). The first one is to build a process map. Second, to perform high-level footprint calculation to help priorities efforts. Following, to collect data across all the life cycle stages. This third step was done by Martínez-Blanco *et al.* (2010). The last step is to calculate the CF.

3. Case study

Both methodologies have been applied to the tomato production in Mediterranean fields. The LCA was applied by Martínez-Blanco *et al.* (2010). This paper completes the study applying the CF methodology to that case.

Two cultivation options are considered depending on whether greenhouse technology was used or not. The two options are: open field cultivation (OP) and greenhouse cultivation (GH).

3.1. Goal and scope definition

This paper aims to study the usefulness of CF tool in the analysis of GWP in agri-food systems and to communicate their environmental performance.

3.1.1. Functional unit

The functional unit chosen is the production of one ton of commercial tomato. All the input and output flows were normalized to this functional unit.

3.1.2. System description

The system analyzed is divided in six stages (Figure 1): Mineral fertilizers production process, transport of Mineral fertilizers to the plots, production and transport of Phytosanitary substances, production and maintenance of Fertirrigation infrastructure, production of Greenhouse infrastructure (only in the GH option) and Cultivation management. In all these stages it was accounted from the production of raw materials to the final disposal of the materials. The six stages will be described in detail in the next section.

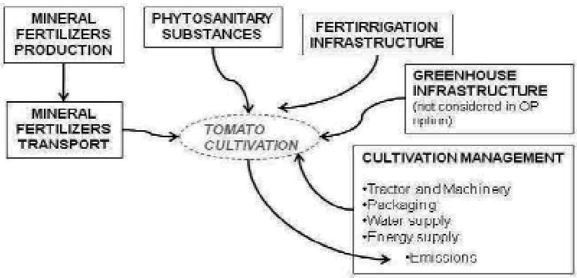

Figure 1: Mediterranean tomato production system

3.1.3. Data origin and quality

Most data used in this study has been collected experimentally in the fields (Martínez-Blanco *et al.*, 2010, Martínez-Blanco *et al.*, 2009). When local information is not available, bibliographical sources and ecoinvent database v2.0 (Swiss centre for life Cycle Inventories, 2007) are used to complete the inventory.

3.1.4. Allocation procedure

During the production process, waste that has a recycling or recovery treatment is not considered on the inventory. It is attributed to the system which uses waste as a raw material. Dumped waste is accounted for (Martínez-Blanco *et al.*, 2010).

3.2. Life cycle inventory

Below a general description is provided. For further details, go to Martínez-Blanco *et al.* (2010).

Mineral fertilizers production includes the whole production process as well as phytosanitary substances production process. For Mineral fertilizers transport from the plant, only outward journey has been assessed because it is supposed that the means of transport return with another load. The dose and types of mineral fertilizers added are different in each cultivation option. They are imported from Israel or Germany depending on the type.

Fertirrigation infrastructure and greenhouse infrastructure stage includes production, transport, installation and waste management of the components. Cultivation management comprises several sub-steps: production and use of all machinery and tools; water and energy supply (accounting for the electricity used by pumps and greenhouse windows movement and also the diesel consumed by the tractor); harvesting boxes and fertirrigation emissions.

3.3. Main differences between CF and LCA in the case study

System boundaries differ depending on the methodology applied (Table 1) largely because CF excludes the emissions arising from the production of capital goods, such as machinery or buildings (Carbon trust *et al.*, 2008), whereas the LCA includes them.

Mineral fertilizers production stage includes consumption of energy and raw materials, waste treatment and emissions generated during this stage for both methodologies. But plant production and its disposal are not included when PAS 2050 is applied. The differences in remaining stages are again due to infrastructures (Table 1).

Table 1: Processes included in CF and LCA methodologies for the stages of the case study

	CF	LCA
MINERAL FERTILIZERS PRODUCTION		
Raw materials production and transport	Yes	Yes
Electricity and diesel	Yes	Yes
Chemical plant and machinery production, maintenance and waste disposal	No	Yes
Emissions	Yes	Yes
MINERAL FERTILIZERS TRANSPORT		
Diesel	Yes	Yes
Lorry and road production, maintenance and waste disposal	No	Yes
Emissions	Yes	Yes
PHYTOSANITARY SUSBSTANCES		
Production and Transport	Yes	Yes
GREENHOUSE and FERTIRRIGATION INFRASTRUCTURE		
Production, construction, maintenance and transport	No	Yes
Waste disposal	No	Yes
CULTIVATION MANAGEMENT		
Diesel, electricity (pump and windows) and irrigation water consumption	Yes	Yes
Tractor and associated machinery production and maintenance	No	Yes
Packaging	Yes	Yes
Fertirrigation emissions	Yes	Yes

4. Results and discussion

Results indicate that there is a relevant difference in GWP depending on which methodology is applied. Table 2 shows the results for each cultivation option and for both methodologies. The differences (in percentage) of quilograms of CO_2 eq. per functional unit between assessments with LCA and with CF are also calculated. For OP there is a difference of 8% in the total kg CO_2 eq. with CF methodology comparing to LCA, meanwhile in the case of GH the difference rises to 27%. The source of variation between both methodologies is the exclusion of infrastructures in CF according to PAS2050.

Table 2: GWP for each cultivation option (GH and OP) considering LCA and CF methodologies

Stages	Greenhouse			Open field		
	LCA	**CF**	**Diff.**	**LCA**	**CF**	**Diff.**
	kg CO_2 eq.	kg CO_2 eq.	%	kg CO_2 eq.	kg CO_2 eq.	%
Mineral fertilizers Production	7,25E+01	6,93E+01	4,5	9,74E+01	9,50E+01	2,5
Mineral fertilizers Transport	6,29E+00	4,97E+00	21,0	7,54E+00	6,02E+00	20,1
Phytosanitary substances	7,21E-03	7,06E-03	2,1	1,47E+00	1,44E+00	2,1
Fertirrigation Infrastructure	3,08E+00	-	100,0	6,05E+00	-	100,0
Greenhouse Infrastructure	3,08E+01	-	100,0	-	-	-
Cultivation Management	3,15E+01	3,12E+01	0,9	2,94E+01	2,86E+01	2,8
TOTAL	**1,44E+02**	**1,05E+02**	**26,9**	**1,42E+02**	**1,31E+02**	**7,6**

On the GH option, 30,8 kg CO_2 eq. associated with the Greenhouse infrastructure are lost when CF is applied (see figure 2). Another infrastructure that produces a relevant variation is the Fertirrigation infrastructure. The latter emits 3,1 kg CO_2 eq. in the GH case and 6,1 kg CO_2 eq. in OP when LCA is applied. Again, both emissions are excluded from CF. In Mineral fertilizers transport there is a reduction of 20% in both cultivation options because of the exclusion of production and maintenance of means of transport.

Cultivation management is the one with fewer differences because it only includes the tractor and associated machinery as capital goods; the rest of elements at this stage are included in both methodologies.

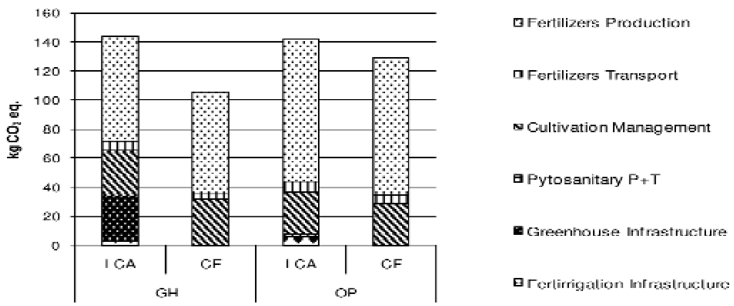

Figure 2: Global warming potential of GH and OP cultivation options applying LCA and CF environmental assessment methodologies

5. Conclusions

The most important difference between use LCA and CF methodologies is the decision of the latter to exclude GHG emissions arising from production of capital goods. The most probable reason is that when performing high-level footprint calculation, these emissions resulting in 1% or lower contribution of the total impact and can be eliminated from the CF calculation. This is true for systems with low infrastructure contribution, as the OP option. In these cases, CF could provide real results. The variability of production processes and the different use of capital goods in agri-food sector mean that they should be included when an agri-food system is studied. By comparing both methodologies it has been shown that the exclusion of capital goods leads a decrease in GHG emissions by up to 30%, giving a misleading result, which is especially important for agricultural products (Frischknecht et al., 2007). Although the current version of PAS 2050 clearly excludes GHG emissions arising from capital goods, it also indicates that these emissions could be included in future revisions (BSI, 2008).

Consequently, if data are available, LCA approach is preferred to analyze agri-food systems because its results are more complete; the whole life cycle is included in the analysis and main impacts are analyzed. Despite all of the above mentioned, CF could be used to complement LCA and serve companies as a decision making measure and communication environmental tool.

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