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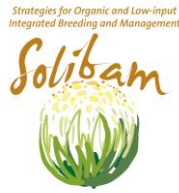
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Dissemination level:

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Abstract

Low-input and diversified cropping systems are sometimes seen as environmentally friendly alternatives to the conventional, high-input agriculture. However, while genetic and species diversities can contribute to the yield stability, the outputs per area are generally lower than standard systems. In this study, we used the methodology of life cycle assessment to evaluate environmental impacts of several diversified cropping systems and compare them to their standard equivalents. The analysis was done per unit area, per product unit and over the whole value chain (from the soil to the fork). In a subsequent study, we developed and applied the methodology of integrative design to reduce environmental impacts of two bread systems from France. Breeding and management innovations were generated in a multi-stakeholder design process and Life Cycle Assessment (LCA) was applied as a decision support tool. Results for four case studies with bread production revealed mostly lower environmental impacts per unit of cultivated area and lower terrestrial and aquatic eco-toxicity both per unit area and per product unit. For the remaining impact categories one out of four case studies performed similarly well or better than the standard systems, while the other farms tended to have similar or higher impacts. The results of the integrative design revealed opportunities for improvements in both systems. Conservative simulation of impacts from improved systems revealed nearly 50% reduction in the global warming potential per kg of bread and 40% in the aquatic eutrophication. The results of this study suggest that there is no direct relationship between the level of inputs or level of diversity and eco-efficiency. Depending on the organisation, in some cases diversified and low-input systems can be relatively efficient, while in others environmental impacts might be much higher than of standard practices. Results of the integrative design exercise revealed, that there are opportunities for large improvements, but the support of quantitative environmental assessment tools might be necessary to support the design process.

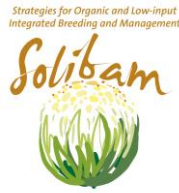


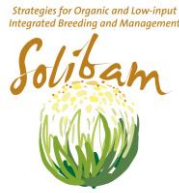
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1 Introduction

Following World War II, Europe has seen large increases in crop yields (FAOSTAT, 2012). This was mainly due to three factors: i.) the increased use of synthetic water soluble fertilisers, ii.) the development and increased use of new varieties with higher yield potential and iii.) the development and application of plant protection products that allowed to keep away pests and diseases. New technologies brought improvements of food security and labour productivity (Broadberry, 2009), but also numerous unintended consequences like burdens on the environment (Stoate *et al.*, 2001). One of the undesired effects was the simultaneous reduction in the number of cultivated crops and varieties (Esquinas-Alcazar, 2005) that can be referred as a reduction of cropping system diversity.

Reduction of genetic and species diversity in agriculture may reduce future resilience of our food system. This is due to the fact that the gene pool available for future breeding purposes is shrinking. Reduction of diversity may also potentially have direct negative environmental impacts. In diversified systems (characterised by high levels of crop and



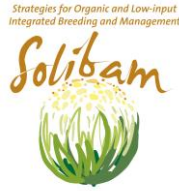
species diversity), rotations of crops prevent the spread of crop-specific pathogens and diseases (Seymour *et al.*, 2012). In homogenous cropping systems (characterised by low levels of crop and genetic diversity), pests are tackled with the use of pesticides, thus potentially causing negative impacts on human health and ecosystems (Hellweg & Geisler, 2003). In diversified cropping systems including legumes nitrogen is fixed for subsequent crops with no external energy input. Without the presence of legumes, synthetic, water soluble nitrogen fertilisers or manures are used to supply nitrogen to the plants. The manufacturing of synthetic, water soluble N fertilisers is associated with the use of natural gas and subsequent high energy demand and carbon dioxide emissions (Patyk & Reinhardt, 1997). The application of nitrogen fertilisers to the soil is also related to the release of nitrous oxide (N₂O), a gas that is per unit of mass 298 times more potent in causing climate change than carbon dioxide.

As a result of the concerns over the negative impacts of homogenous cropping systems, there has been some renewed interest in diversified cropping systems. In Italy, there is a growing consumer demand for products made of ancient varieties, landraces or even wheat ancestors, such as emmer or spelt (Guarda *et al.*, 2004; Piergiovanni, 2013). In France, there are now 69 active associations of farmers who view maintaining genetic heterogeneity as an important part of cropping systems (Réseau Semences Paysannes, 2012). These farmers cultivate mixtures of varieties or cereals in rotations and often use traditional methods of farming, such as animal labour (PROMMATA, 2013) to reduce the external energy input. Products of diversified cropping systems may not always comply with the requirements of modern processing and retail industries who demand large volumes of products of uniform quality. Therefore some farmers and consumers organise alternative processing and distribution channels, referred in the literature as “alternative food networks” (Renting *et al.*, 2003).

The empirical evidence over the comparative environmental advantage of diversified systems over standard methods of production is still lacking. This report describes result of research that aimed to address this gap. Under the term “diversified cropping systems” we understand cultivation of variety mixtures or heterogeneous ancient cultivars of cereals or cultivation of more than one species of vegetables on each 1 ha of farmland. Life Cycle Assessment (LCA) methodology was applied in the study. The method allows for systematic quantification of environmental impacts of products, services, processes or systems (Finnveden *et al.*, 2009). Its key characteristic is the consideration of all relevant substance flows from the moment of their extraction from nature to the point of their release to the environment. It considers the whole product life cycle and the broad range of environmental impacts thus allowing for a systematic assessment of environmental advantages and disadvantages of different systems. The study described in this report had two main goals: 1.) To compare the environmental impacts of cropping systems with high levels of diversity (genetic and species) to standard methods of production and 2.) To quantify the potential for improvements of environmental performance through innovative breeding and management interventions, further referred as SOLIBAM strategies.

2 Methodology

The two goals of the study were addressed through an approach consisting of two methods. To address the first goal of the study, we conducted a comparative LCA of products from diversified cropping systems and their standard equivalents. In order to address the second



objective, we developed and applied a methodology combining benefits of integrative design process (Charnley *et al.*, 2011) and LCA.

2.1 Part 1. Comparative LCA of diversified cropping systems and their standard equivalents

In this part of the study, environmental impacts of six diversified cropping systems located in two different European climatic zones (Temperate Oceanic and Mediterranean) were assessed with the use of LCA. Results for case study cropping systems were compared to four standard references representing average practices of farmers in the respective regions of Europe.

2.1.1 Description of systems under study

The selection of case study farms covered two different European climatic zones: Temperate Oceanic and Mediterranean as well as two contrasting scales of production: farms below 10 ha and above 70 ha. Selected producers aimed at minimisation of external inputs at the agricultural stage as a strategy for improving environmental performance. The farmers strived for high diversity in the production systems by cultivating variety mixtures or heterogeneous ancient cultivars and a high crop diversity. All the processing and distribution occurred on-farm or within the distance of 50 km. More details on the selection on the case study farms can be found in SOLIBAM deliverable D8.8 (Wright *et al.*, 2014).

Fig. 1 shows the approximate locations and key information about analysed case study cropping systems as well as references to which these systems were compared. Table 1 provides characteristics of the studied cereal-based cropping systems.

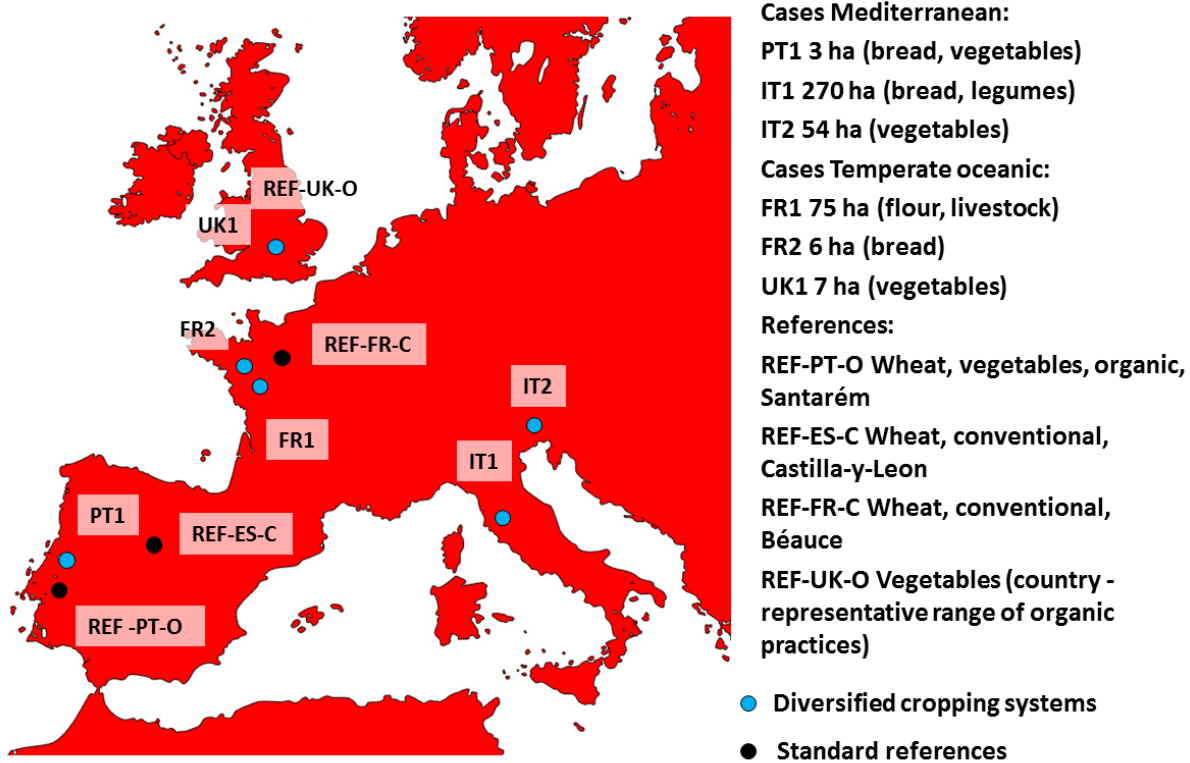


Fig. 1. Locations and key characteristics of the studied cropping systems

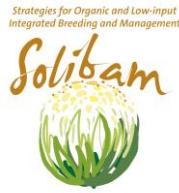


Table 1. Key characteristics of the studied cereal-based cropping systems

	FR-1	FR-2	IT1	PT1	REF FR C	REF ES C	REF PT O
Surface [ha]	75	6	270	3	unspecified	unspecified	125 ha total 20 ha organic 105 ha conventional
Climate	Temperate oceanic	Temperate oceanic	Mediterranean	Mediterranean	Temperate oceanic	Mediterranean	Mediterranean
Crop rotation:	5 years grassland with legumes, rye, winter wheat	winter wheat, winter rye, intercropped barley-peas	chickpeas, winter wheat/einkorn green manure, millet/oat	Potatoes, Brassicas, Fabaceae, Alliums, winter wheat, rye, oatmeal, green manure	wheat after cereals	sunflower-winter wheat-winter barley-spring barley	winter wheat, winter barley, tomato, broccoli
Fertilization	Organic, Solid composted cattle manure 10 t ha ⁻¹ yr ⁻¹ (74 kg N, 39 kg P ₂ O ₅)	Organic, Solid composted horse manure 12 t ha ⁻¹ yr ⁻¹ (65 kg N, 30 kg P ₂ O ₅)	Organic, Plant residues and small quantities of products 0.3 t ha ⁻¹ yr ⁻¹ (36 kg N, 48 kg P ₂ O ₅)	Organic, Solid sheep manure (10 kg N ha ⁻¹ , 3 kg P ₂ O ₅ ha ⁻¹)	Synthetic, (190 kg N ha ⁻¹ 43 kg P ₂ O ₅ ha ⁻¹)	Synthetic, (57 kg N ha ⁻¹ 47 kg P ₂ O ₅ ha ⁻¹)	Organic, Solid cattle manure (249-272 kg N, 32-140 kg P ₂ O ₅)
Pesticides	No pesticides input	No pesticide input	Seed propagation with copper oxychloride 1.89 g kg ⁻¹ seed	Bacillus thuringiensis, av. 0.3 application yr ⁻¹	Pesticides, av. 6.5 applications yr ⁻¹	Pesticides, av. 1.5 application yr ⁻¹	Bacillus thuringiensis, av. 1 application yr ⁻¹
Yield	1.3 - 1.5 t ha ⁻¹	0.6 - 2.3 t ha ⁻¹	0.7 - 1.5 t ha ⁻¹	1 - 1.4 t ha ⁻¹	7.5 t ha ⁻¹	2.9 t ha ⁻¹	5 t ha ⁻¹

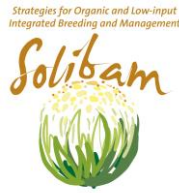
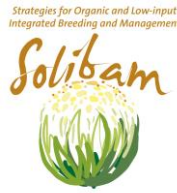


Table 2: Key characteristics of the analysed vegetable systems.

	UK1	IT2	REF-UK _{high} and REF-UK _{low}	REF-PT-O
Cropping systems	48 different vegetable species on less than 6 ha of organic land 7 year crop rotation in the fields, 9 year crop rotation in the garden Greenhouse and two polytunnels Stockfree™ organic standard (no animal inputs)	14 vegetable species on less than 6 ha of organic land Vegetables and legumes (ryegrass, vetch, peas, oat, broad bean) Greenhouse and open fields irrigated with the use of tractor	Modelled based on the average practices of organic farmers in the UK. The system REF-UK _{high} represents the highest value in the range of agricultural inputs and yields. The system REF-UK _{low} represents the lowest value in the range of agricultural inputs and yields.	20 ha of organic land with tomato and broccoli in the rotation with wheat and barley
Climate	Temperate oceanic	Mediterranean	Temperate oceanic	Mediterranean
Fertilisation	Legumes and wood chip compost	Manure, cover crops and phosphate rock	Manure and phosphate rock	Manure and phosphate rock
Crop protection	No spraying. Flower strips and beetlebanks designed to encourage functional biodiversity	The use of plant protection products permitted in organic farming	The use of plant protection products permitted in organic farming	The use of plant protection products permitted in organic farming
Distribution	Box scheme	Direct distribution and a box scheme	Large retailers	Supplying processing plant (producing canned tomatoes) and large retailers

Table 3: Comparison of 3 UK organic vegetable supply systems

	UK1	UK2		UK2			
	areas	Low-input reference		High-input reference			
	(ha)	Yields ^a (t/ha)	areas (ha)	Output (t)	Yields ^a (t/ha)	areas (ha)	Output (t)
Potatoes, Early		10	0.42	4.2	20	0.21	4.2
Potatoes, main crop		15	0.83	12.45	40	0.31	12.45
Carrots		15	0.41	6.15	50	0.12	6.15
Cabbage, white		20	0.26	5.2	50	0.11	5.2
Cauliflower		16	0.23	2.99	24	0.15	2.99
Parsnips		10	0.21	2.1	30	0.07	2.1
Beetroots		10	0.35	3.5	30	0.12	3.5
Onions		10	0.35	3.5	25	0.14	3.5
Leeks		6	0.48	2.88	18	0.16	2.88
Squash		15	0.17	2.55	40	0.06	2.55
Courgettes		7	0.88	6.16	13	0.47	6.16
Lettuce,		6	1.73	10.38	9.6	1.08	10.38
Total, vegetables	4.02	9.91	6.26	62.06	20.62	3.01	62.06
Green Manure	1.56		1.58 ^b			0.75 ^b	
Field margins, infrastructure	0.78		1.12 ^c			0.53 ^c	
Total area	6.36		9.02			4.29	



2.1.2 Goal and scope definition

The goal of the LCA was to compare environmental impacts of diversified cropping systems to their standard equivalents. Cropping systems are multifunctional. To address the goal of the study, two system boundaries and four functional units were used:

2.1.2.1 System boundary: analysis at the farm gate

The system boundary at the farm gate is most common in agricultural LCA, thus allowing comparison of results to other studies. The analysis covers all inputs and processes that were required to deliver the storable product at the farm gate. Two functional units were used for the assessment at the farm gate:

- a) 1 ha of land occupied during one year (reflecting the land management function of cropping systems, i.e. the impacts related to cultivating 1 ha of agricultural land)
- b) 1 tonne of product at the farm gate in the form it is used for sale or further processing (reflecting the productive function)

2.1.2.2 System boundary: analysis at the level of the whole value chain

This system boundary reflects differences in impacts over the whole product life cycle (Fig. 2). Muñoz *et al.* (2010) distinguished 6 stages in the life cycle of foods that have important impacts on the environment and should be considered in LCA: Food production (including agriculture), wholesale and retail, home processes, kitchen waste management, human excretion and wastewater treatment. We assumed that environmental impacts during the stages human excretion and wastewater management will be the same for products from diversified systems and standard equivalents. These stages were therefore outside the scope of the analysis. For the analysis of cereal-based cropping systems, we have included baking and distribution. This is due to the fact that these processes differed substantially between standard supply chains and the alternatives, largely as a result of farmer's decision to cultivate mixtures of cereal varieties. Harvested mixtures of grains cannot be sold to industrial millers and bakers and these farmers organised alternative ways of processing and distribution. Vegetable farmers UK1 and IT2 are distributing their products through a box scheme on the limited, local market. Cooking was outside the system boundary, assuming that raw vegetables from both analysed supply chains will require the same energy and materials for cooking. The functional units used in this analysis were:

- c) 1 kg of bread ready for the consumption at the consumer's home

Or

- d) a defined quantity of raw vegetables at the consumer's home (the quantity is given in the specific sections)

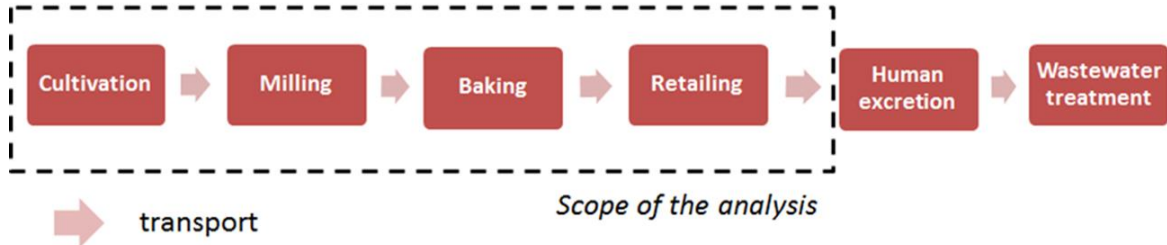
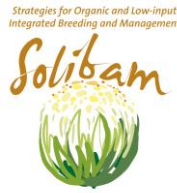


Fig. 2. Stages in the life cycle of bread with negative impacts on the environment. (Own work based on Muñoz et al. 2010)

2.1.3 Sources of data

Primary data for the analysis of case studies were collected directly from cereal and vegetable farmers in Portugal, Italy, France and the United Kingdom. Researchers from Agroscope and RISØ DTU visited case study farms and collected the data in collaboration with local partners. Data collection was done through direct semi-structured interviews on farm and further contact via e-mail, telephone and post. Three years of data were covered: 2008, 2009 and 2010 at most case study farms with the exception of UK1. Consideration of several years was needed to reflect the variability of results due to random events that may occur from year to year, such as extreme weather conditions leading to reduced or failed harvest. In one case, UK1, only two years of data were covered (i.e. 2009 and 2010). The selection of cases represents diversified both cereal and vegetable cropping systems in two different climatic zones: Mediterranean and Temperate oceanic. Swiss Agricultural Life Cycle Assessment (SALCA) models (Gaillard & Nemecek, 2009) were used to estimate direct field emissions at the cropping system level. Unless stated otherwise, life cycle inventories were derived from the ecoinvent database v2.2 (Hischer *et al.*, 2010). Emissions from burning diesel were modelled based on the inventories for farming operations from the ecoinvent database. Regional adjustments of important life cycle inventories were made. This included changing electricity mixes from Swiss to the country-specific conditions and adaptations to the material use in capital goods to better reflect the situation on farm – in particular the wooden building at FR2 and irrigation pipeline in UK1. Due to the lack of inventory for the production of mills and the on-farm oven for bread production, a standard inventory for agricultural machinery was used, using the mass figures given by the farmer. The life cycle inventory for vegetable seedlings was constructed based on the study of Stoessel *et al.* (2012) and for biomulch based on Patel *et al.* (2003). Emissions during the composting of woodchips were derived from Wihersaari (2005).

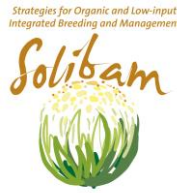
The data representing high-input organic cropping system from Portugal (REF-PT-O) were collected directly from the farmer. The data for the conventional wheat cropping system in Spain were supplied by the ITA research institute as a representation of average practices of farmers in the region Castilla-y-Leon (von Richthofen *et al.*, 2006). Data representing average practices of conventional wheat farmers in Béauce region of France were supplied by the Eure-et-Loire region's Chamber of Agriculture (UNIP, 2011). Data related to the processing and distribution of products from diversified cropping systems were collected directly from



farmers. For the preparation of bread at home, we assumed that customers of FR1 will bake two standard loaves of bread 750 g each in a domestic electric oven for half an hour what corresponds to most recipes found in cookbooks. The energy use of an electric oven was derived from the specifications of the European Council Directive 92/75/EEC for a medium-sized household electric oven of the median energy efficiency class (D). Distances from the farm to the industrial mill were assumed to be the same as an average haulage distance for products of the categories “agriculture, hunting and forestry as well as for fish and other fishing products”. This was taken from the database of the Directorate-General of the European Commission (Eurostat, 2013) for each country and year considered. Similarly, distances between the industrial mill and the industrial bakery as well as between the industrial bakery and the retail store were assumed to be the same as the average transport distances for food products, beverages and tobacco and taken from the same database. Life cycle inventories for industrial flour making and industrial bread making were derived from the Danish LCA Food database (Nielsen & Nielsen, 2003a, b). Although the use of salt was not reported in the Danish study, we have added 10g of salt per kg of bread to ensure consistency with other studied breads. Energy use in the supermarket and space requirements for bread were taken from the study of Tassou *et al.* (2009). We assumed that the bread will be displayed at room temperature for 24 hours and that 5% of the mass will be wasted and sent to the landfill. Distances between the consumer’s home and the supermarket were taken from the survey of Rizet & Keita (2005). We assumed a petrol-engine passenger vehicle (European average) to model resource use and emissions during the shopping trip. The standard production and distribution of organic vegetables in the UK was modelled based on the data from the survey of farms in the UK (Lampkin, 2011) supported by the opinion of researchers from the Organic Research Centre. Two model systems UK2low and UK2high represent the standard range of organic practices in the UK. The system UK2low represents standard practices with the minimum amount of farm external inputs per area and the lowest yields while UK2high represents practices with the highest amount of farm-external inputs per area and the highest yields. Details on crop management and yields in both models are provided in Table 2 and can also be found in a peer-reviewed publication (Markussen *et al.*, 2014).

2.1.4 Allocation procedures

Allocation in LCA is the process of dividing environmental burdens from the production system to different co-products. Following the ISO 14040 standards, allocation was avoided whenever possible. One situation in which it was not possible was the division between the grains and straw in case of the farm FR1. The farmer cultivates varieties of cereals in which the ratio of grain to the total biomass is 0.5 (harvest index). The straw is exported from the cropping system and is utilised as an input for livestock production. ISO 14040 recommends to allocate the impacts to the co-products based on the underlying physical relationships between them. This principle of physical causality was inadequate in this situation, as the functions of two co-products – grains and straw – have very different uses and therefore a common physical causality is hard to defend. Therefore, we applied the economic allocation to divide environmental impacts between the grains and the straw. The prices of wheat and rye were derived from FAO statistics, and the price of straw from farmer interviews. Depending on the year in question, this yielded allocation factors between 0.29 and 0.35 for straw. It is worth mentioning, that mass allocation was tested in a sensitivity analysis. Due to



the relatively high harvest index, this yielded exclusions of more emissions from the study scope for this farm (50%) and did not affect any conclusions.

Allocation was also needed to estimate environmental impacts from transportation of products. Mass allocation was applied to account for transport emissions when bread or vegetables were transported together with other items. This choice was made as there is a direct relationship between the weight of the product and the fuel use.

In the second part of the study (integrative design), capital goods such as farm buildings were included. Allocation choices for farm buildings were made according to the function of the building. Grain storage facilities were allocated to different grains based on the volume taken up for storage. In a rare case where buildings were serving multiple purposes on farm (such as the warehouse for the machinery serving multiple uses, such as tractors) we applied economic allocation.

Upstream environmental impacts related to the production of manure or woodchips were not assigned to the analysed cropping systems (outside the system boundary). This was to follow a cut-off approach making a clear division between systems producing waste by-product and systems using them. According to this criterion, emissions from livestock farming should be assigned to the main products of livestock farming – meat and milk – and not the manure. Similarly, emissions from gardening should be assigned to the function of maintaining the garden not providing wood chips for the farmer UK1. However, all the environmental impacts from the transport of inputs, their storage and composting on farm were included.

2.1.5 Life Cycle Impact Assessment (LCIA)

The following impact assessment methods were applied in the analysis:

- *Non-renewable energy demand* as derived from oil, natural gas, uranium, coal and lignite, also referred to as a cumulative energy demand (Frischknecht *et al.*, 2007).
This impact assessment method considers the use of energy in non-renewable energy resources (fossil and nuclear) throughout the whole life cycle of products, processes and systems. This includes energy consumed during the extraction of raw materials, refining, manufacturing and disposal. All raw resources are traced to the moment of their extraction from nature. Following the calculation of quantities of all raw materials, characterisation factors are applied based on their gross calorific value. Results of impact assessment are expressed in Megajoules.
- *Global Warming Potential* over 100 years GWP₁₀₀ based on IPCC (IPCC, 2007)
Climate change presents one of the biggest environmental problems of modern society, this impact category is therefore among the most commonly used in LCA. The IPCC method traces all greenhouse gasses that are directly and indirectly released into the atmosphere as a result of the analysed product life cycle, service or activity. Characterisation factors for different gasses are based on the information from IPCC (IPCC, 2007). The method allows to choose different time horizons for assessing the impact on climate change. In this analysis we looked at the impacts over the 100 years timeframe.
- *Photochemical ozone formation* (vegetation), further referred as ozone formation, according to EDIP2003 (Hauschild and Potting 2004)
Substances such as volatile organic compounds (VOCs), nitrogen oxides (NO_x), carbon monoxide (CO) and methane (CH₄) have a capacity to catalyse the formation



of ozone in the troposphere under the exposure to sunlight, the process called photochemical ozone formation. In the lower parts of the atmosphere, ozone reacts with organic compounds, posing damage to various living organisms, especially plants. In this method, all emissions to air are included and characterization factors are calculated based on the potential exposure of vegetation to ozone above the safe threshold levels for plants. The impact is expressed as the area of vegetation that is potentially exposed to the threshold of chronic effects (over 40 ppb) in square meters, the annual duration of the exposure in hours and the concentration above the threshold as expressed in parts per billion.

- *Eutrophication potential* to aquatic ecosystems (further referred as aquatic eutrophication N) according to EDIP2003 (Hauschild & Potting, 2004)
The overload of nutrients in aquatic ecosystems can lead to unwanted consequences, including anoxia. Aquatic eutrophication N can be caused by all emissions to air, water and soil that contain biologically available nitrogen. The characterisation factors in this method are calculated with the use of the CARMEN (Cause and effect Relation Model to support Environmental Negotiations) model. The spatially explicit model simulates the transport of nutrients to surface water from agricultural supply, through groundwater drainage, surface run-off and atmospheric deposition. The factors derived from the model express the fraction of a nutrient emission from the soil or wastewater treatment plant that will reach and expose surface waters. Results of this impact category are expressed in nitrogen equivalents (N-eq) and reflect the maximum exposure of aquatic systems to the emission.
- *Acidification potential* according to EDIP2003 (Hauschild and Potting 2004)
The deposition of acidifying substances leads to a decrease of pH in soils and water bodies. This is mainly caused by ammonia, sulphur oxides, and nitrous oxides (NO_x). The characterisation factors for acidification are calculated based on the potential of a given substance to release hydrogen ions. The patterns of deposition are calculated with the use of the previously mentioned RAINS model. Results of impact assessment are expressed as the area of ecosystem within the deposition area that will exceed the threshold critical load for acidification.
- *Aquatic eco-toxicity, terrestrial eco-toxicity and human toxicity potentials* according to CML01 (Guinée *et al.*, 2006).
For the evaluation of toxicity-related impacts, CML01 methods from the Center of Environmental Science of Leiden University were applied. The characterisation of impacts in the CML method is based on the modelling of fate, exposure and effect of toxic substances on “areas of protection”, such as terrestrial or freshwater ecosystems or human health. The method employs the model USES 2.0 developed at the Dutch National Institute for Public Health and the Environment (RIVM). The impact category “freshwater aquatic eco-toxicity” from the CML method is further referred in the deliverable as “aquatic eco-toxicity” and covers all impacts on freshwater aquatic ecosystems. The method takes into account all emissions to air, water and soil as determined in the life cycle inventories. The impact on ecosystems is calculated as the ratio of predicted environmental concentration to the predicted no-effect concentration. The impact category “terrestrial eco-toxicity describes effects on terrestrial ecosystems. Similarly to aquatic eco-toxicity, the impact is calculated as the ratio of predicted environmental concentration to the predicted no-effect



concentration. The impact category “human toxicity” covers impacts on human health. The effect is derived as the ratio of acceptable daily intake to the predicted daily intake of the given substance. Results in all these three toxicity –related impact categories are derived as kilograms of 1,4-dichlorobenzene equivalents. For the pesticide active ingredients the characterisation factors calculated by Hayer *et al.* (2010) were used.

2.2 Part 2. Integrative design

In this part of the study, the potentials for improvements at two case study farms was assessed with the use of integrative design methodology. Fig. 3 explains the conceptual framework of the applied methodology.



Fig. 3. Methodological framework of the applied integrative design procedure

To minimise model uncertainty, farmers were given the opportunity to provide feedback on the constructed LCA model. Environmental hot-spots and opportunities for improvements were also discussed. In a subsequent step, results of LCA were presented to the consortium of experts during an interdisciplinary workshop. Experts included agronomists and plant breeders, representatives of seed companies and farmer associations. Results of LCA were accompanied by the detailed description of studied systems that included farm size, location, soil and cropping system management characteristics as well as methods of processing and product distribution. 21 experts were divided into 5 working groups. At least one scientist in each group had previously visited the farm and had the knowledge of its key limiting factors. Workshop participants were presented with cards containing a range of potential innovations for improving the environmental performance. The list was compiled based on the extensive review of literature on strategies for improving eco-efficiency of low-input cropping systems (Kulak *et al.*, 2013). Some cards were left blank to encourage participants to develop own ideas. For each selected improvement option, participants were asked to provide qualitative information on the relative cost of strategy implementation and potential yield improvements. Each workshop session was followed by explanations of participants behind the rationale of their choices and a plenary discussion. Results of the workshop were consulted with the concerned producers. Each strategy proposed by expert groups was discussed with the farmer one by one in subsequent semi-structured face-to-face interviews. Solutions that were rejected by the farmer, for various reasons, were not considered in further analysis. Producers were



also encouraged to provide their own opportunities for improving eco-efficiency or propose other scenarios to be analysed with LCA. The consultation resulted in establishing the list of potential management interventions and LCA models were applied to simulate their influence on the environmental impacts.

2.3 Critical Review

The LCAs in this report were carried out according to the ISO standards 14040 and 14044 (ISO, 2006a, b). A formal critical review according to these ISO standards was not performed. However, all LCAs performed in this project except for one vegetable case from Italy are subject to an external scientific review process. Results of LCA for vegetables from the UK were published in the peer-reviewed journal *Sustainability* (Markussen *et al.*, 2014). Results for bread are currently undergoing a review for publication in a peer-reviewed journal with the focus on LCA studies.

3 Results Part 1: Comparative LCA of products from diversified cropping systems and their standard equivalents

3.1 Cereals and bread

3.1.1 Agricultural stage (grains)

Results for all the considered environmental impacts at the agricultural stage were highly variable. Some of the considered cases of diversified cropping systems, namely FR1 and PT1, were characterised by similar or lower cumulative energy demand per product unit to standard references while FR2 and IT1 provided higher impacts (Fig. 4). Rye had lower non-renewable energy resource use than wheat in the case FR2, while the opposite result was found for the case in Portugal (PT1). Results were also characterised by high variability from year to year, especially in the case of the farm FR2. For the functional unit 1 ha of land (Fig. 5), results for cumulative energy demand were clearly higher in reference scenarios. This is due to the higher use of agricultural inputs per ha. The production of synthetic, water soluble fertilisers that were considered to be applied in Spanish and French systems is energy intensive (Patyk & Reinhardt, 1997). In the case of Portuguese high-input organic system, impacts on cumulative energy demand were caused i.e. by relatively intensive soil preparation and the production of potassium chloride.

Similar trends to non-renewable energy demand could be observed for the global warming potential (Fig. 6 and Fig. 7) and ozone formation (Fig. 8 and Fig. 9). These impact categories were highly affected by diesel use for farming operations and the yield. The Portuguese high-input organic farm had higher impact on the global warming potential than the Spanish reference per ha, while the opposite was found for the cumulative energy demand. This is because organic N fertilisers, such as a manure, are by-products of a livestock production and therefore do not require significant input of energy during their production. However, their application to the field is followed by the release of nitrous oxide which is a potent greenhouse gas. The French farm FR1 had lower impacts on the global warming potential and ozone formation than FR2 per unit of product, but higher per unit of land. This is due to the fact that the methods of cultivation employed by FR2 are much more extensive as additional



land is needed for the production of feed for horses and the production of hay and barley-peas requires less inputs per ha than wheat.

Results for aquatic eutrophication N revealed slightly different trends. Both per product unit (Fig. 10) and per unit of area (Fig. 11), organic farms had higher impacts on this category than conventional farms due to the use of manure. Per unit of area, the organic high-input producer (REF-PT-O) had significantly higher impacts than the rest of the systems, but this was not the case per unit of product as the farm is characterised by relatively high yields. The lower-yielding wheat from the Italian case and the wheat from the case FR2 were characterised by the highest impacts on aquatic eutrophication N per product unit. Very high slopes in the case of Italian farm increased the risk of surface run-off and erosion. In France, additional fertilised land was needed to produce feed for horses. For acidification, the diversified systems FR1 and PT1 had lower impacts than the other farms per product unit (Fig. 12). Per unit of area (Fig. 13) standard references were characterised by clearly higher impacts. This impact category is highly sensitive to direct airborne emissions especially of ammonia and nitrous oxide, which are dependent on the amount of applied fertilisers. For FR1, the good result per product unit was partially owed to low fertilisation and partially the integration of crop and livestock production that allows to make best use of farmyard manure and to valorise the by-product of crop production – straw. For PT1, the advantage was mainly due to the low amounts of applied fertilisers.

The impact category human toxicity appears to show similar trends (Fig. 14 and Fig. 15), but the causality is different here. In particular, the direct relationship to fertilisation is weaker as this impact category is sensitive to the emissions of other substances, such as polycyclic aromatic hydrocarbons and heavy metals. Interestingly, the high-input organic producer from Portugal (Ref-PT-O) showed higher human toxicity impact per area than conventional farming in France or Spain. This is mainly due to the use of sulphur-based pesticide. The aquatic ecotoxicity was much higher in one case – the conventional wheat production in France than the rest, both per unit of agricultural area (Fig. 16) and per product unit (Fig. 17). This is due to the use of synthetic herbicides in the French cropping system, in particular the synthetic herbicide containing chlorotoluron. The impact on terrestrial ecotoxicity was also dominated by the French conventional case (Fig. 18 and Fig. 19) with the smaller difference to the case FR2 in case of results per product unit. Relatively high eco-toxicity of FR2 per product unit was caused by emissions of heavy metals. Small quantities of heavy metals are both embodied in the machinery and released to the soil during diverse farming operations, which – in respect of the low yields – results in relatively high impacts per tonne of cereals.

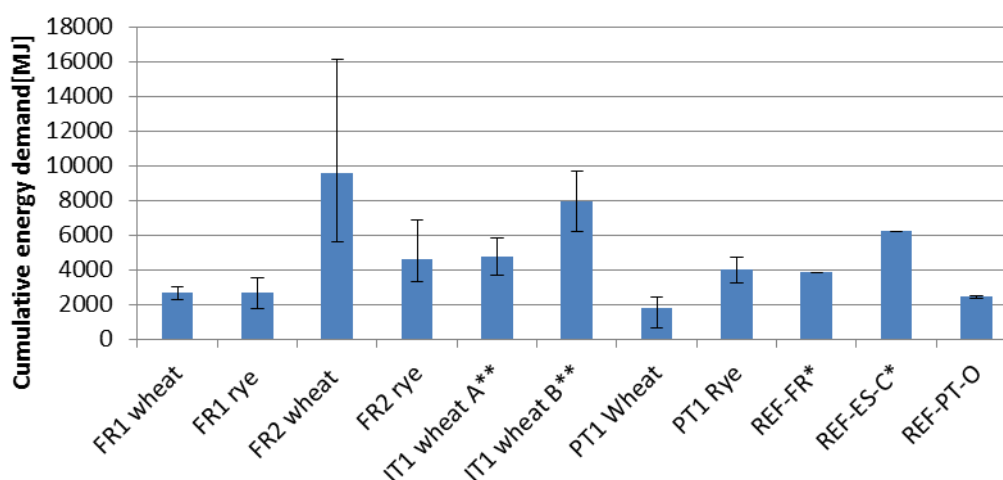


Fig. 4. Results at the agricultural stage for non-renewable energy demand (Cumulative energy demand). Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

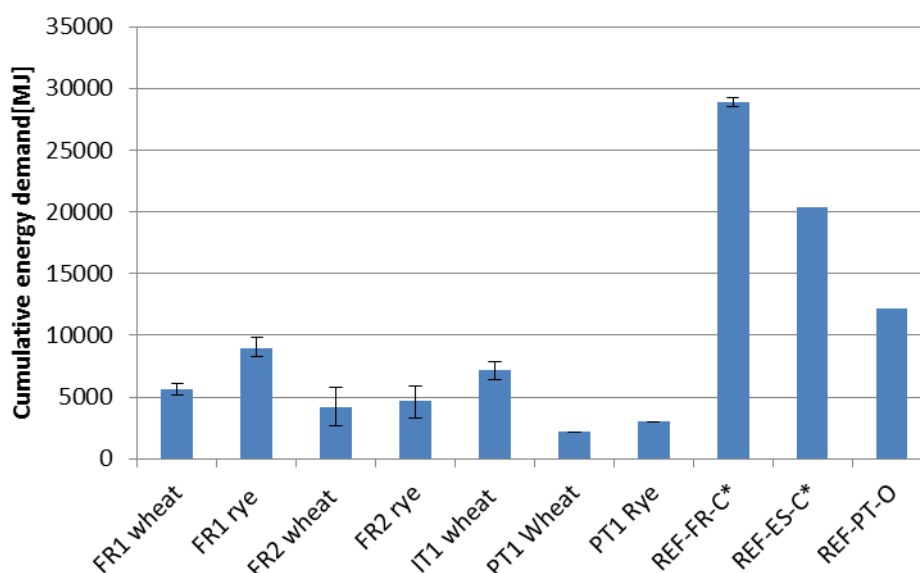


Fig. 5. Results at the agricultural stage for non-renewable energy demand (Cumulative energy demand). Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

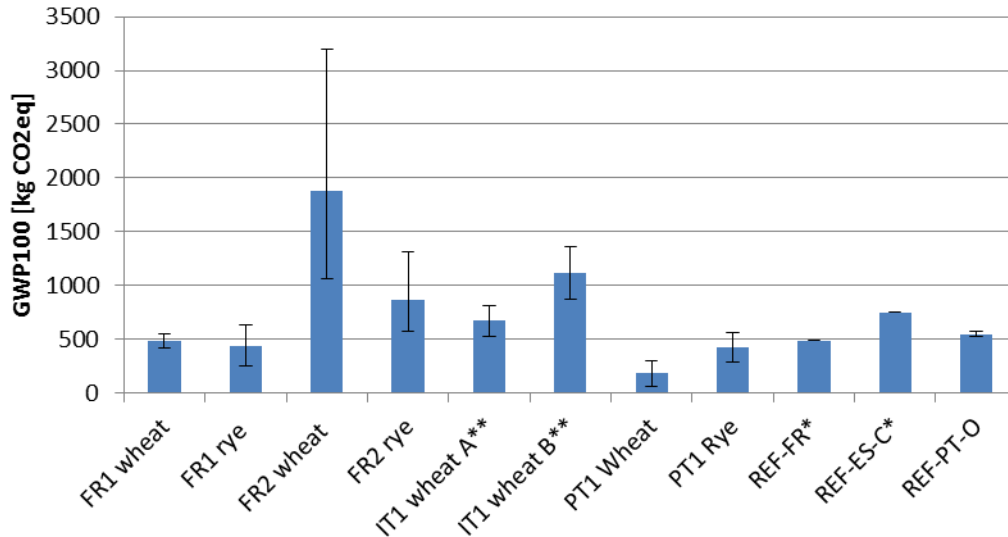


Fig. 6. Results at the agricultural stage for Global Warming Potential over 100 years. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

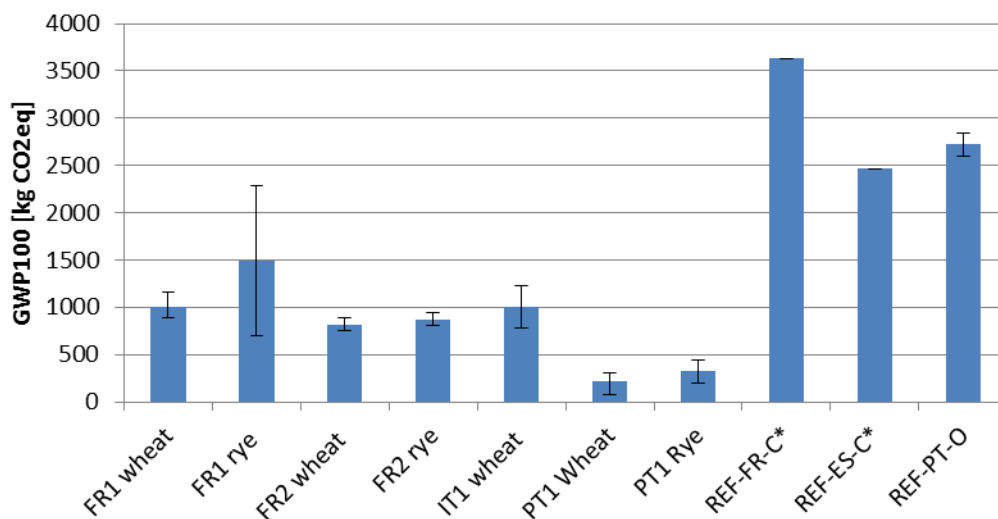


Fig. 7. Results at the agricultural stage for Global Warming Potential over 100 years. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

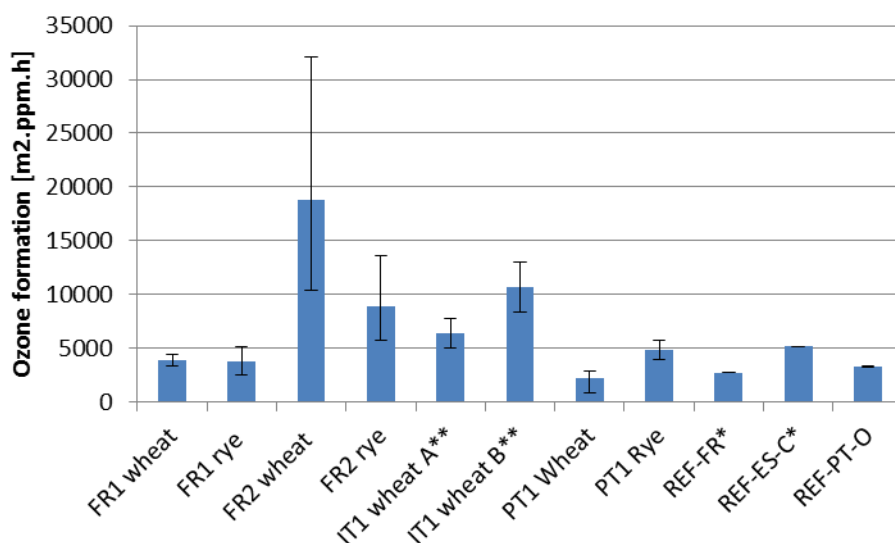


Fig. 8. Results at the agricultural stage for Ozone formation. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

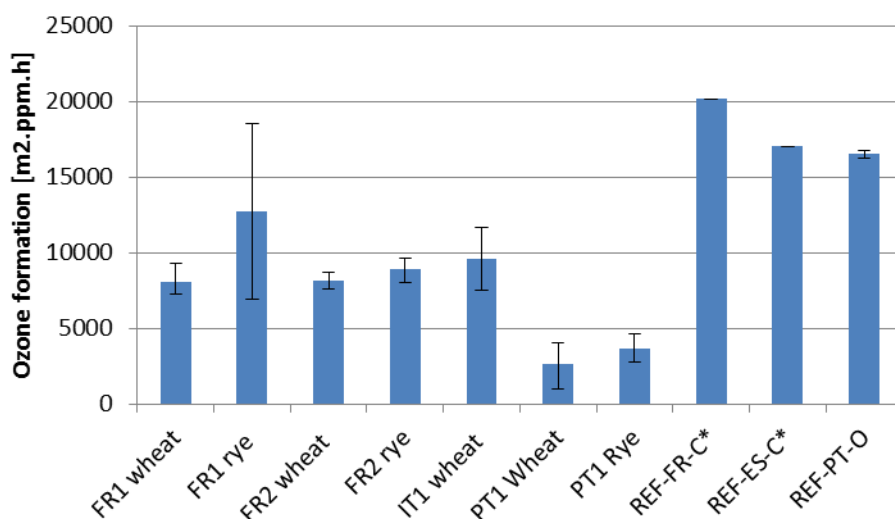


Fig. 9. Results at the agricultural stage for Ozone formation. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

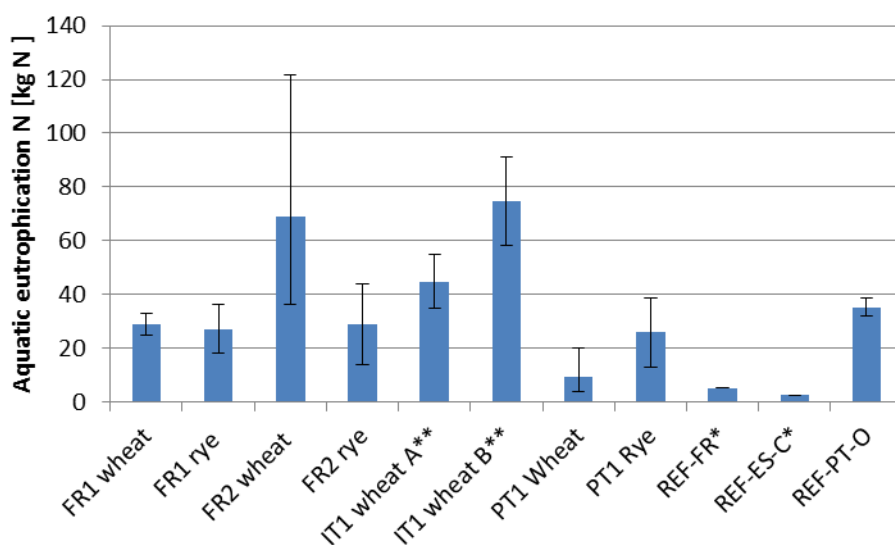


Fig. 10. Results at the agricultural stage for Aquatic eutrophication potential N. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettabile*.

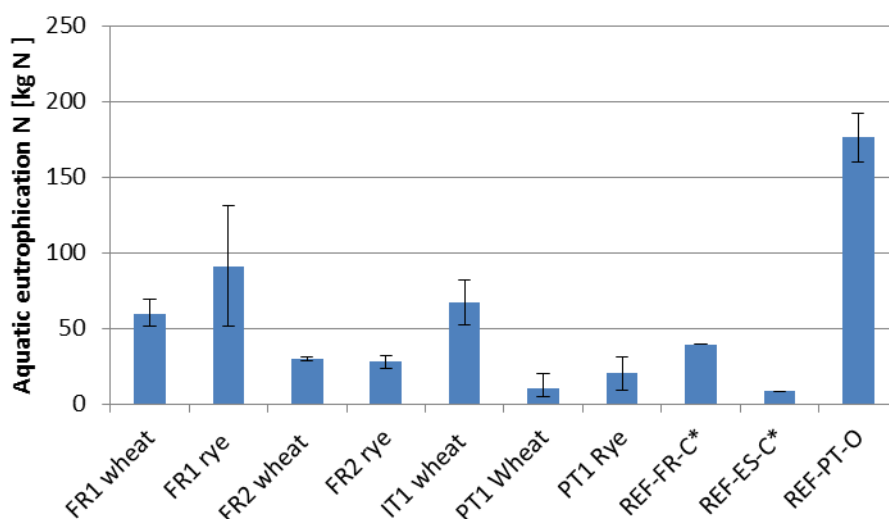


Fig. 11. Results at the agricultural stage for Aquatic eutrophication potential N. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

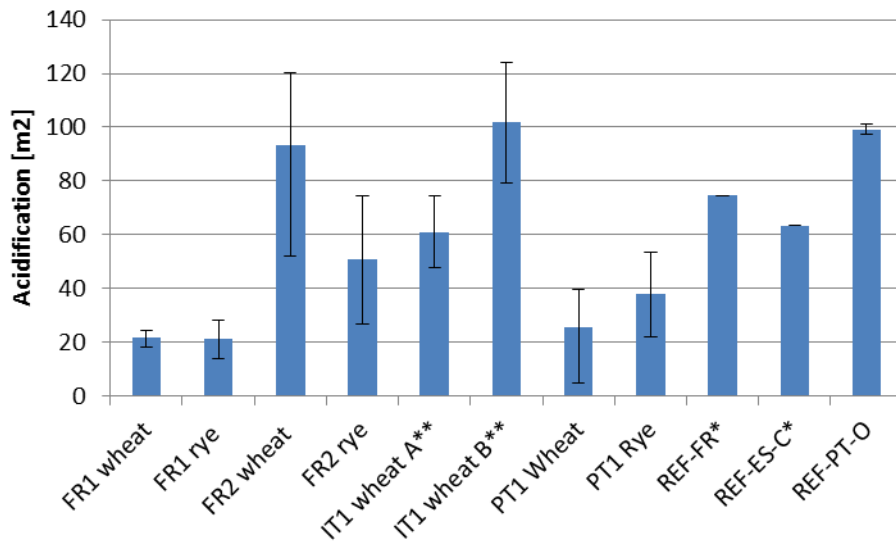


Fig. 12. Results at the agricultural stage for the Acidification Potential. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

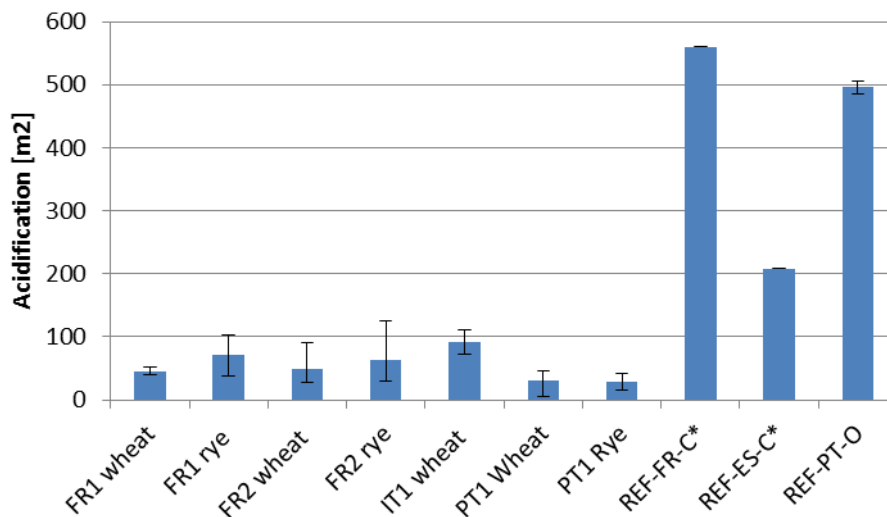


Fig. 13. Results at the agricultural stage for the Acidification Potential. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

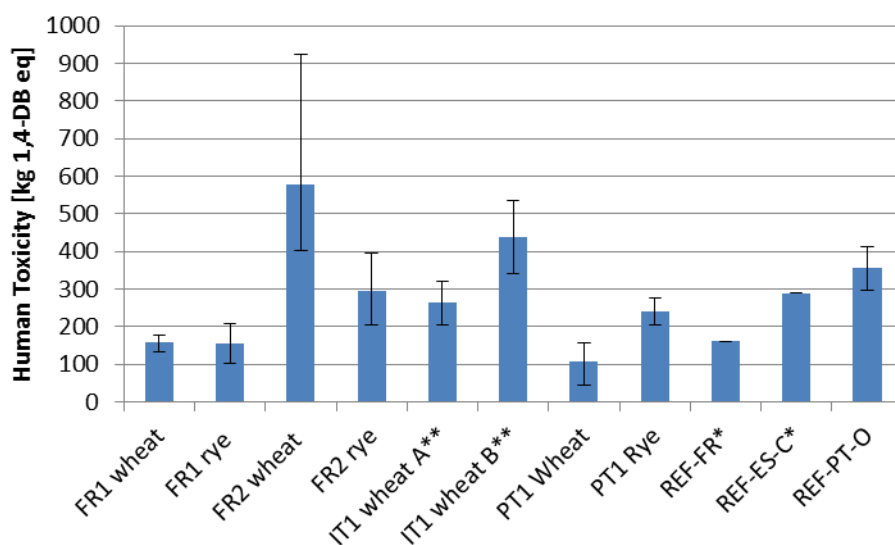


Fig. 14. Results at the agricultural stage for Human toxicity. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso, Frassineto, Verna, Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

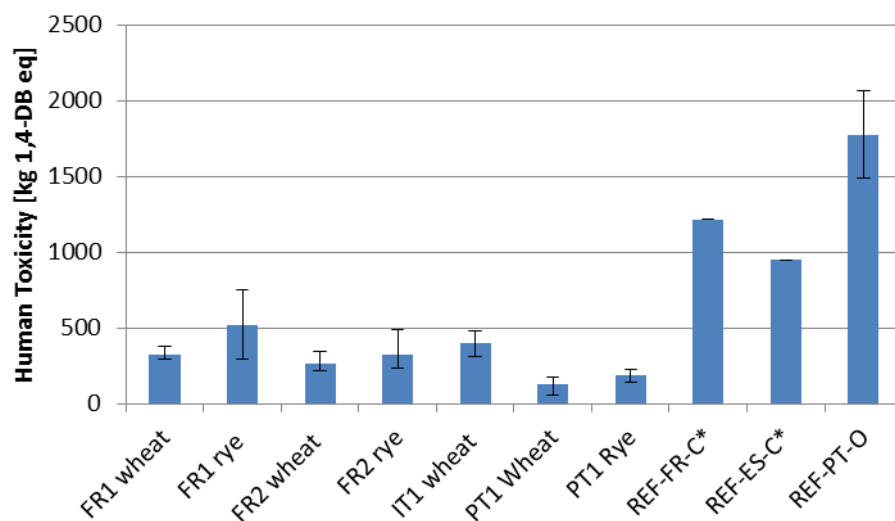


Fig. 15. Results at the agricultural stage for Human toxicity. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

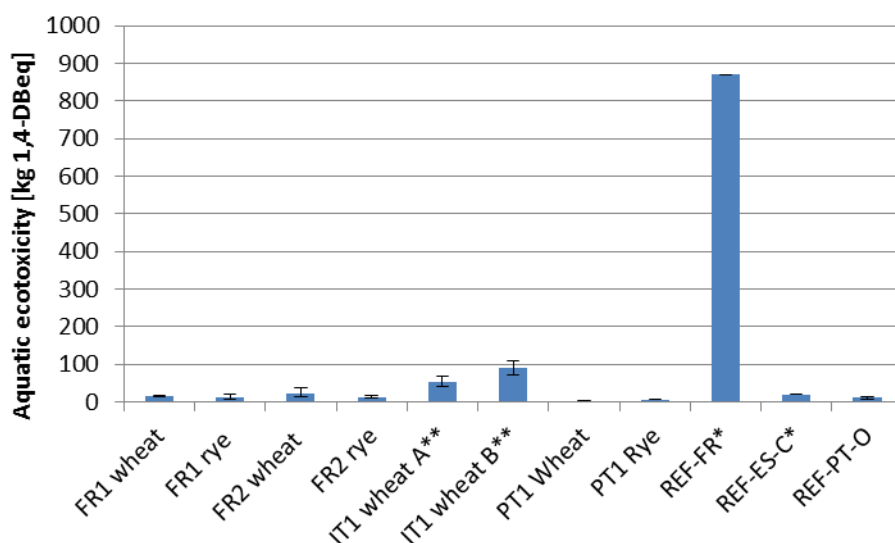


Fig. 16. Results at the agricultural stage for the Aquatic eco-toxicity potential. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

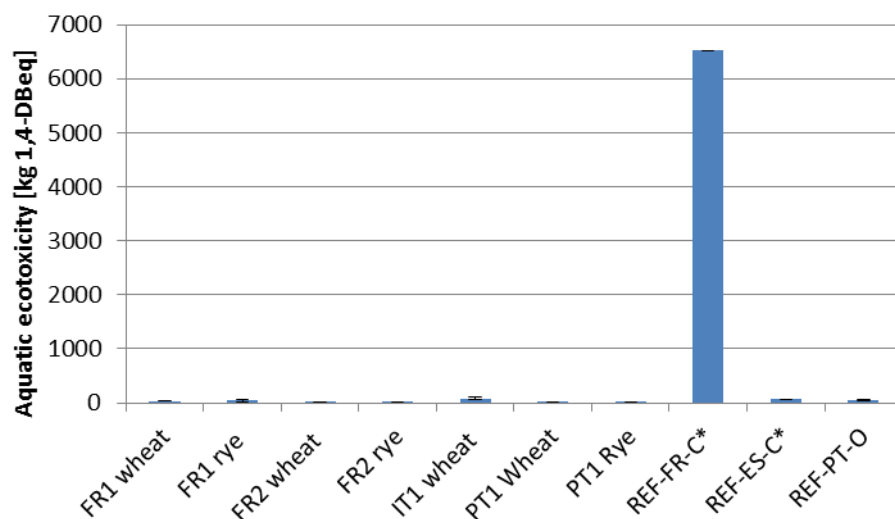


Fig. 17. Results at the agricultural stage for the Aquatic eco-toxicity potential. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in management. * information on variability not available.

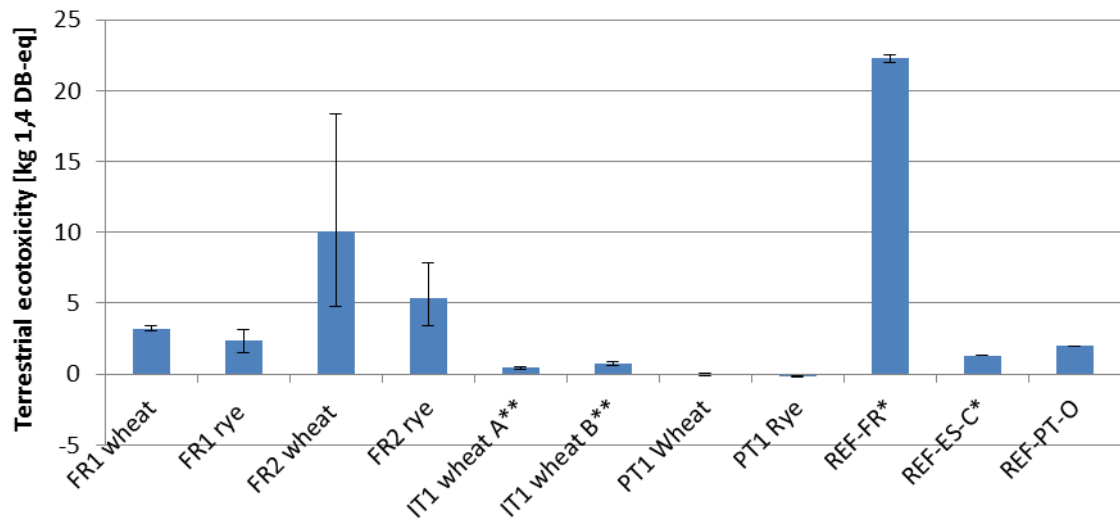


Fig. 18. Results at the agricultural stage for the Terrestrial eco-toxicity potential. Functional Unit: 1 tonne of cereals at the farm gate. Error bars represent yearly variability due to changes in yields and management. * information on variability not available. ** Wheat A – higher yielding cultivars: *Gentil Rosso*, *Frassineto*, *Verna*, *Abbondanza*; Wheat B – lower yielding cultivar: *Inallettibile*.

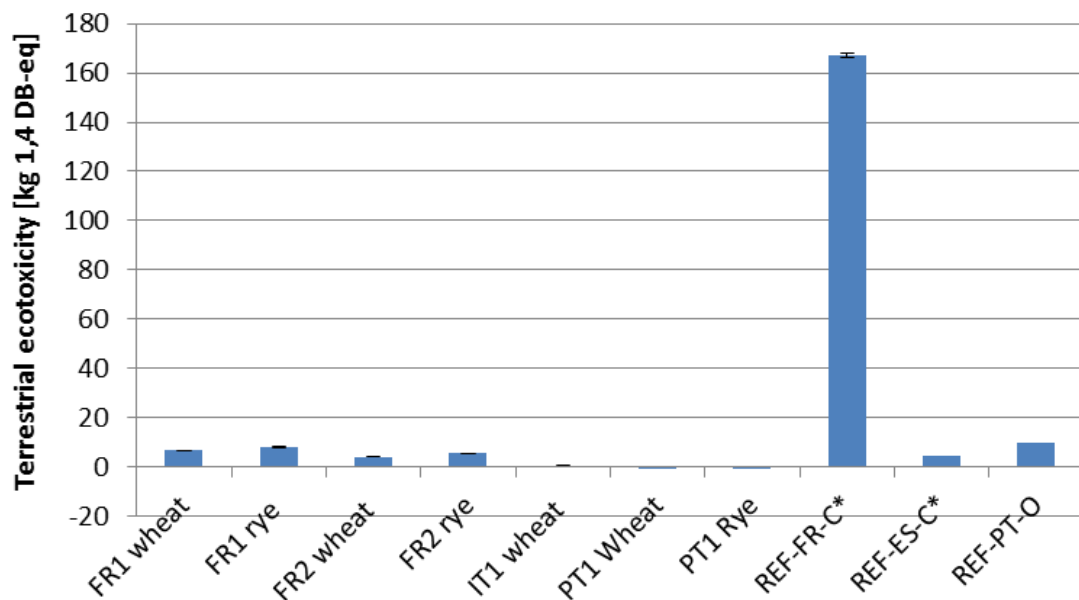
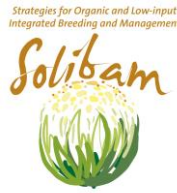


Fig. 19. Results at the agricultural stage for the Terrestrial eco-toxicity potential. Functional Unit: 1 ha of land occupied during one year to produce cereals. Error bars represent yearly variability due to changes in yields. * information on variability not available.



3.1.2 Whole value chain (bread at consumer's home)

Results for the food supply systems were characterised by even larger variability. The diversified case FR1 revealed a similar impact on non-renewable energy demand to the standard references, while the remaining cases of diversified systems scored higher. The impact on cumulative energy demand at the level of the whole value chains was similar or slightly higher in diversified systems as compared to the references (Fig. 20). The advantage of FR1 at the agricultural stage was largely offset here by the relatively high impact of baking. This is due to the fact that the farmer sells flour and not bread and consumers bake the bread at home. Home-baking is more electricity intensive than industrial baking. The electricity mix in France contains a large share of nuclear energy and therefore requires uranium.

The trend was different for the global warming potential (Fig. 21), where French cases scored better than the other ones. Due to the high share of nuclear energy, the French electricity mix is characterised by high use of non-renewable energy resources per kWh but low global warming potential compared to other countries. The case FR2 had a high contribution of the agricultural stage to the global warming potential. This is partially owed to the use of draft horses for some of the farming operations. Horses cause lower use of non-renewable energy resource use as compared to mechanical traction, but require additional land for the production of feedstuff.

The trend in impact on ozone formation (Fig. 22) was similar to the global warming potential, with the difference that cases FR2 and IT1 revealed high impact of baking. This is due to the fact that the baking at these farms is done with the use of wood and wood combustion causes release of nitrous oxides.

Post-agricultural stages of the product life cycle revealed little to no impacts on aquatic eutrophication N (Fig. 23), which is highly correlated with the use of fertilisers on farm, especially manure.

The farm FR1 had lower impacts on acidification than the remaining cases (Fig. 24). This impact category is affected by a number of factors. One of them is the organisation of the distribution; fuel burning during distribution increased the impacts of PT1. The method of baking can also affect acidification. Baking with wood at the farms FR2, and IT1 had relatively high impacts on acidification due to the emission of nitrogen oxides and sulphur dioxide. The country's electricity mix was another important factor. Countries with a large shares of electricity derived from coal have high per kWh impact on acidification, because of the emissions of nitrogen oxides and sulphur dioxide during coal burning.

A slightly different trend was observed for human toxicity (Fig. 25). The relative contribution of baking with wood was high to this impact category, mainly due to the emission of polycyclic aromatic hydrocarbons during wood combustion. The bread from the farm FR2 had the highest impact on this impact category, partially due to baking with wood and partially due to the combustion of fuel for farming and distribution. The relative impacts on terrestrial and aquatic ecotoxicity were not affected by the inclusion of post-agricultural stages (Fig. 26 and Fig. 27) because these two impact categories are largely dependent on the emission of pesticides and heavy metals on farm. In both cases, the French reference was still responsible for the largest impact due to the use of herbicides.

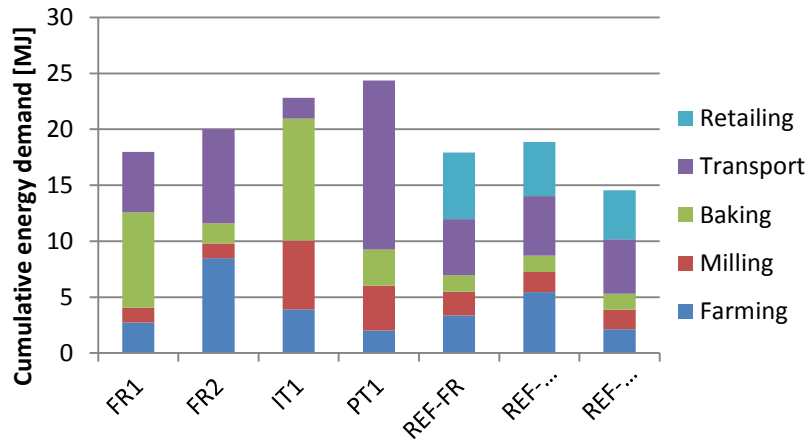


Fig. 20. Results over the whole value chain for the non-renewable energy resource use. Functional Unit (FU): 1 kg of bread at the consumer's home.

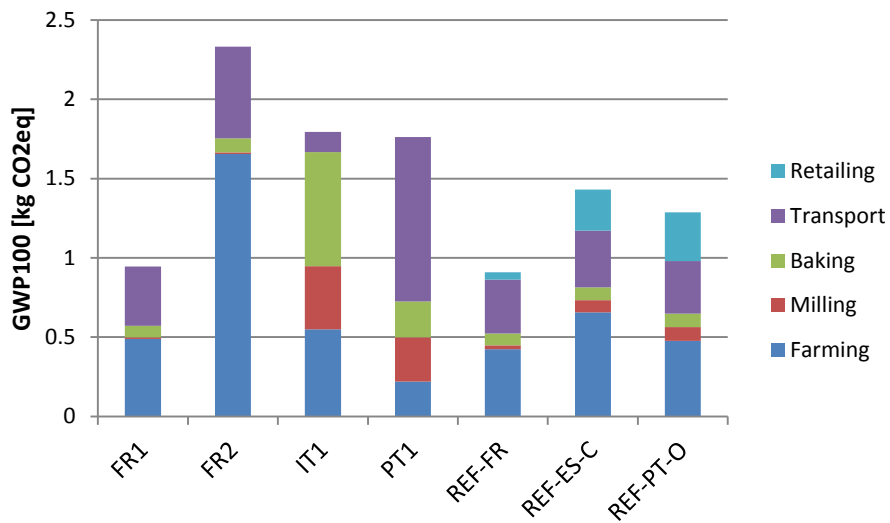


Fig. 21. Results over the whole value chain for the Global warming potential over 100 years. Functional Unit: 1 kg of bread at the consumer's home.

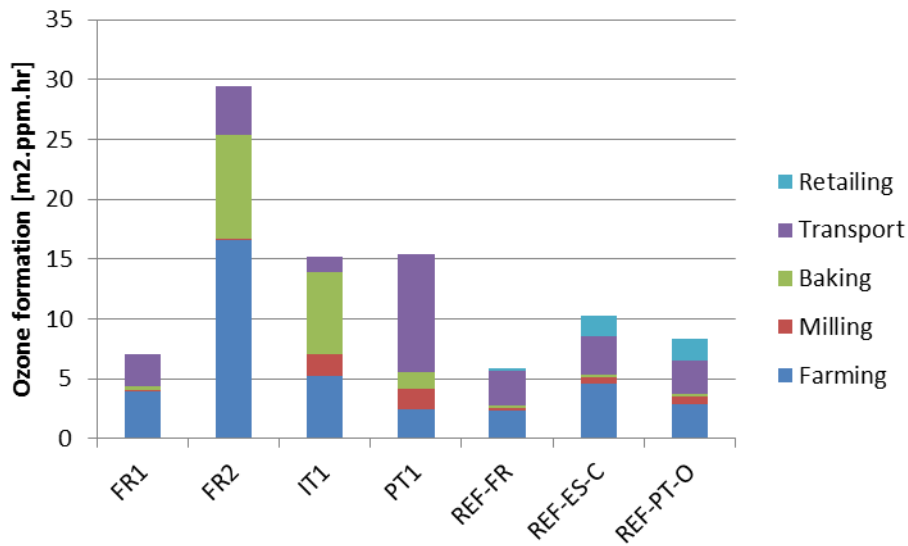


Fig. 22. Results over the whole value chain for the Ozone formation potential. Functional Unit: 1 kg of bread at the consumer's home.

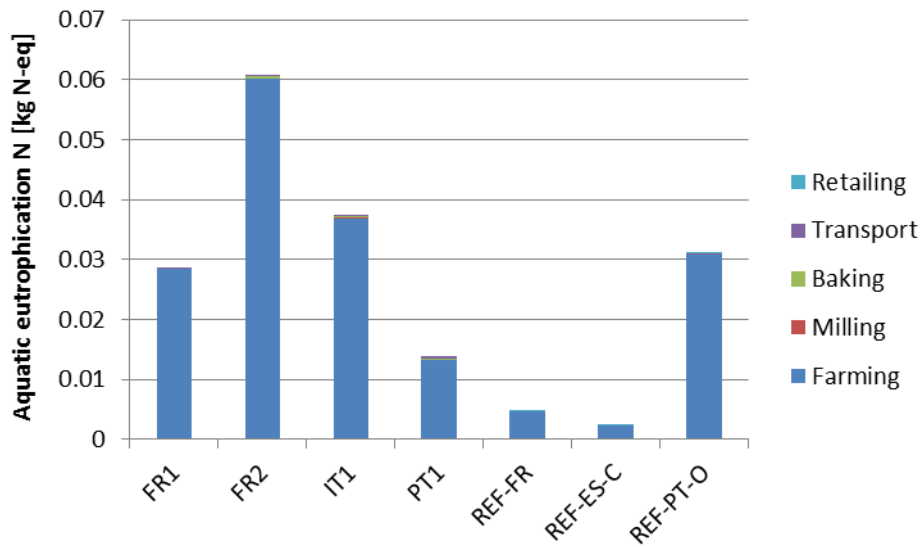


Fig. 23. Results over the whole value chain for the Aquatic eutrophication potential N. Functional Unit: 1 kg of bread at the consumer's home.

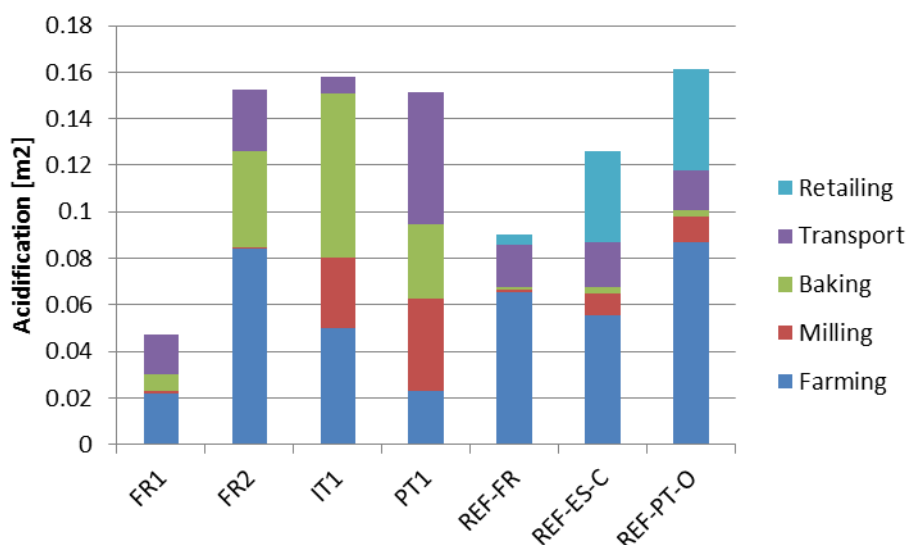


Fig. 24. Results over the whole value chain for the Acidification potential. Functional Unit: 1 kg of bread at the consumer's home.

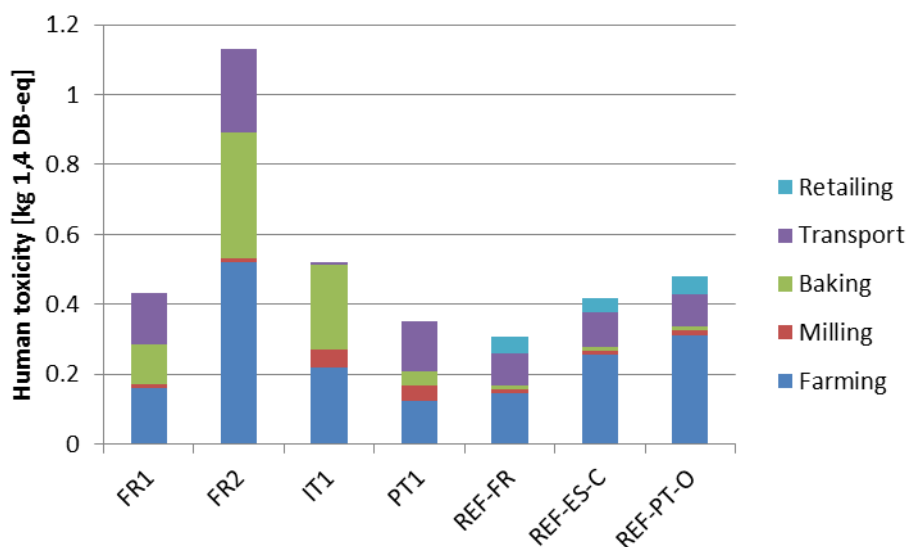


Fig. 25. Results over the whole value chain for the Human toxicity potential. Functional Unit: 1 kg of bread at the consumer's home.

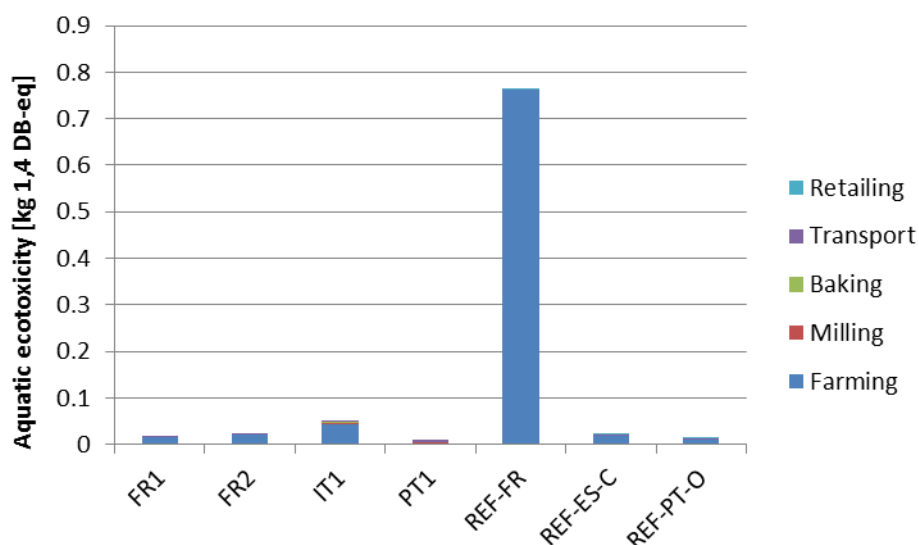


Fig. 26. Results over the whole value chain for the Aquatic ecotoxicity potential. Functional Unit: 1 kg of bread at the consumer's home.

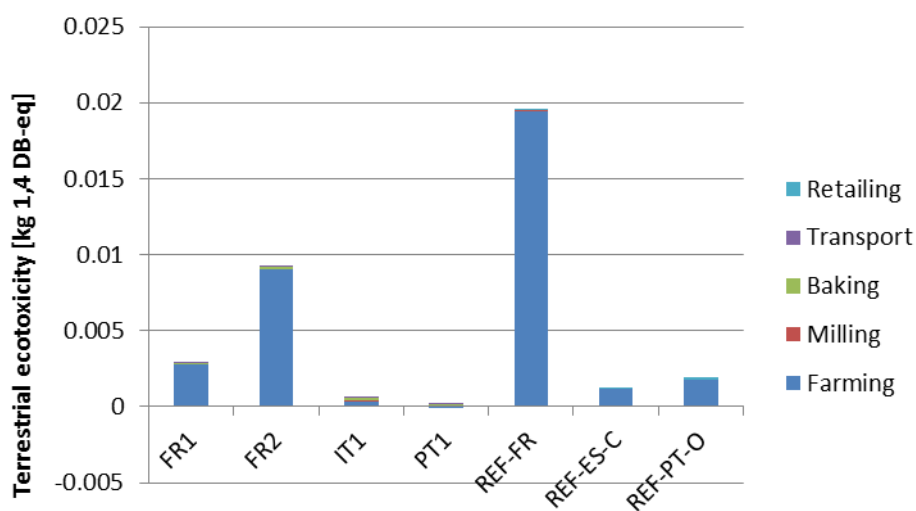
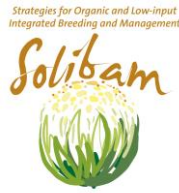


Fig. 27. Results over the whole value chain for the Terrestrial ecotoxicity potential. Functional Unit: 1 kg of bread at the consumer's home.

3.2 Vegetables

The first part of this chapter presents results for potatoes. This crop was chosen because it allows a comparison across different cases. It is also an important crop from the dietary perspective and constitutes a significant portion of sales at all analysed vegetable farms. The second part of this section provides result for the vegetable case study from UK. Here, a specific mix of vegetables from a case study was compared to the range of organic practices in the UK.



3.2.1 Agricultural stage: Potatoes

The LCA results for potatoes from IT2 were characterised by higher cumulative energy demand than both PT1 and the range of references from UK (Fig. 28). The relatively high impact was largely caused by irrigation. The latter process is energy and resource intensive. IT2 also revealed the highest impact on the global warming potential (Fig. 29), but unlike in the case of non-renewable energy resource use, the Portuguese case had the second highest impact. The impacts on ozone formation showed similar trend to the cumulative energy demand although with the smaller relative difference (Fig. 30). The situation was different for aquatic eutrophication N that was much higher in case of the organic reference from Portugal due to high amounts of applied manure (Fig. 31). Human toxicity (Fig. 32), terrestrial ecotoxicity (Fig. 33) and aquatic eco-toxicity (Fig. 34) were higher in the diversified Italian case than the rest. This was due to the large amounts of applied pesticides that are permitted in organic farming, especially rotenoids.

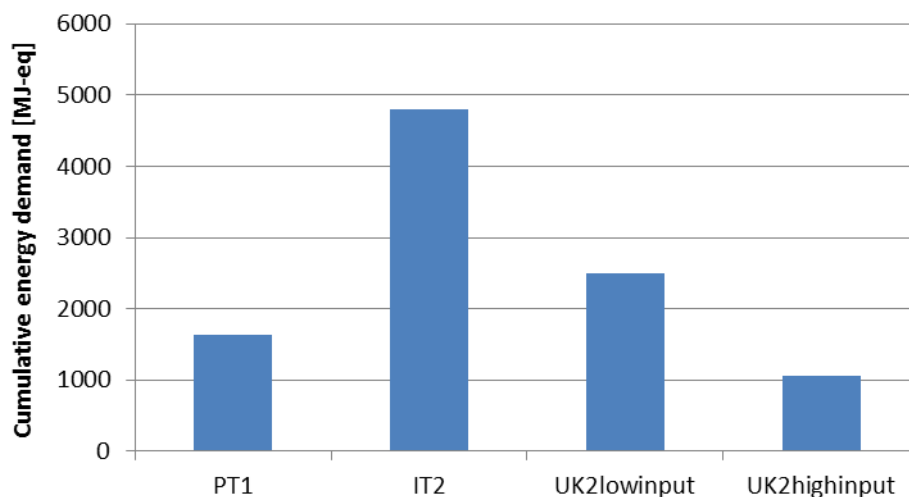


Fig. 28. Results at the agricultural stage for the non-renewable energy demand. Functional Unit: 1 tonne of potatoes at the farm gate.

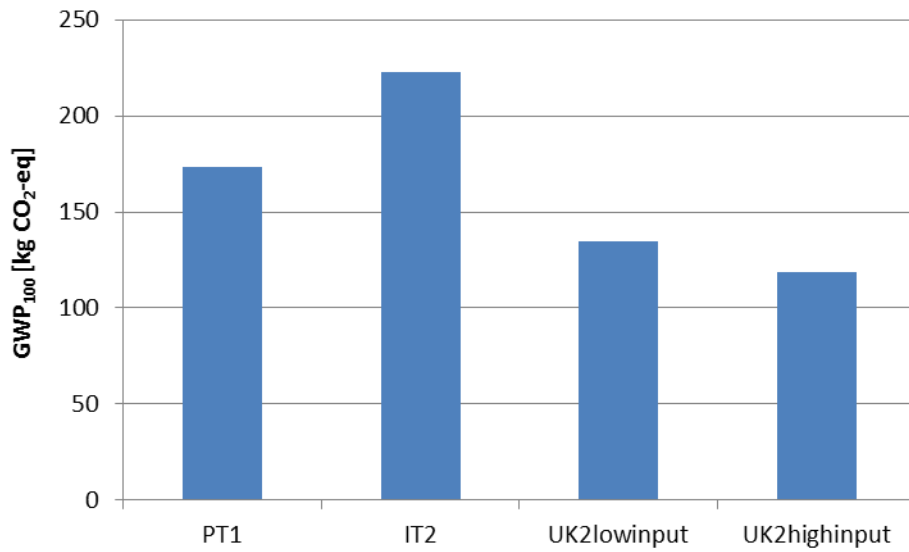


Fig. 29. Results at the agricultural stage for the Global warming potential over 100 years. Functional Unit: 1 tonne of potatoes at the farm gate.

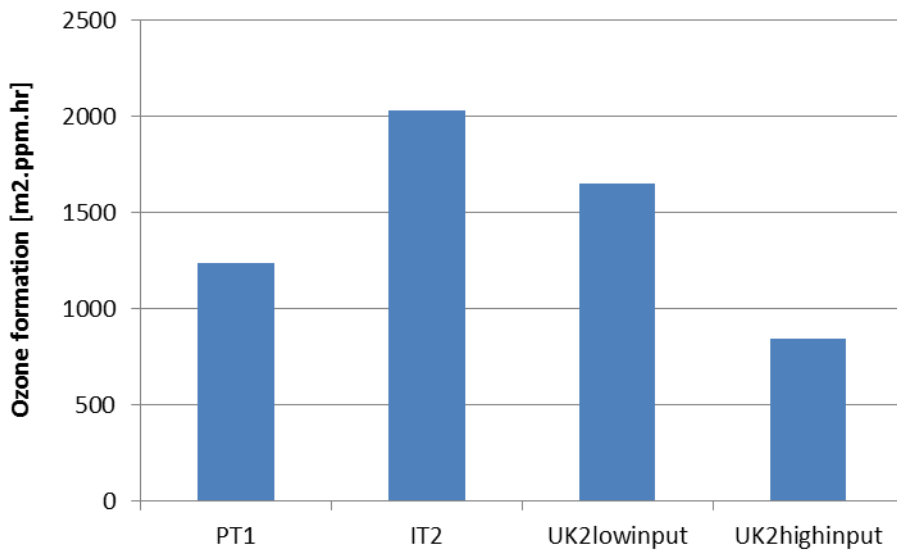


Fig. 30. Results at the agricultural stage for the Ozone formation potential. Functional Unit: 1 tonne of potatoes at the farm gate.

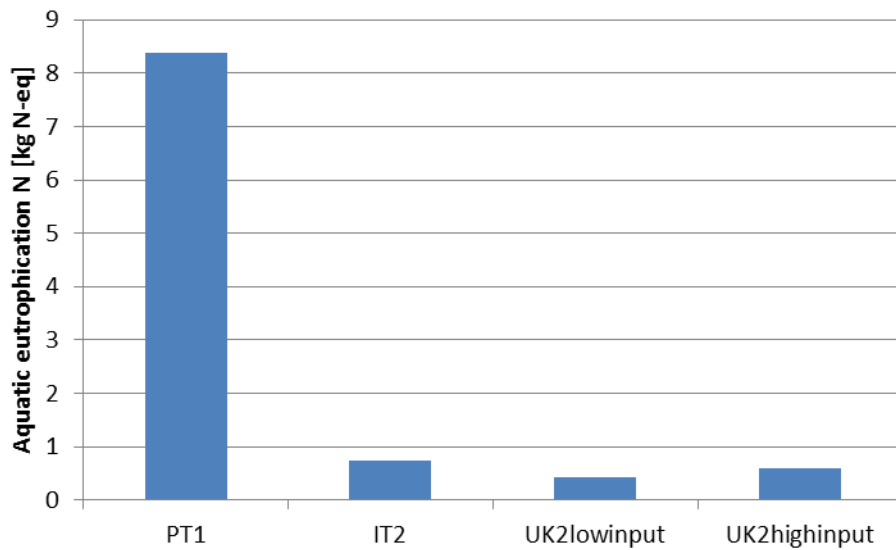


Fig. 31. Results at the agricultural stage for the Aquatic eutrophication potential N. Functional Unit: 1 tonne of potatoes at the farm gate.

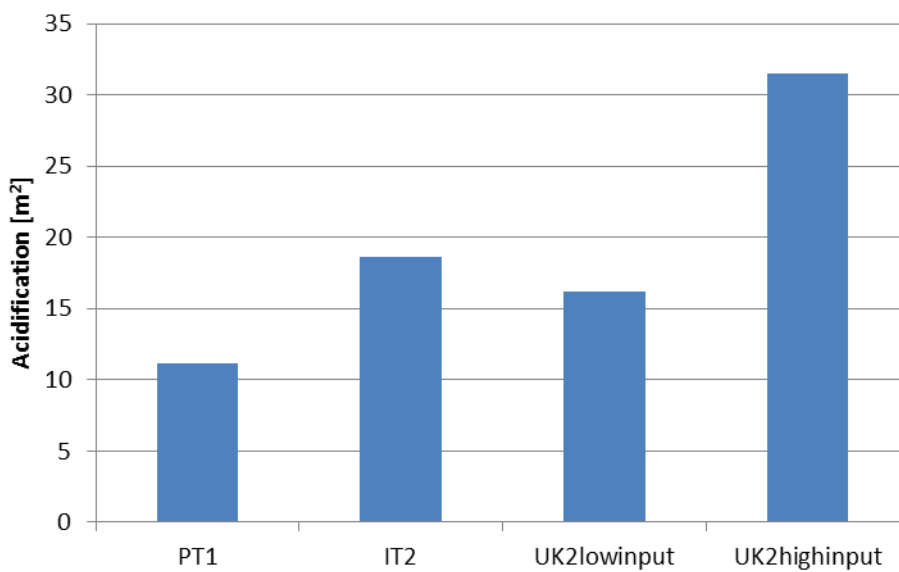


Fig. 31. Results at the agricultural stage for the Acidification potential. Functional Unit: 1 tonne of potatoes at the farm gate.

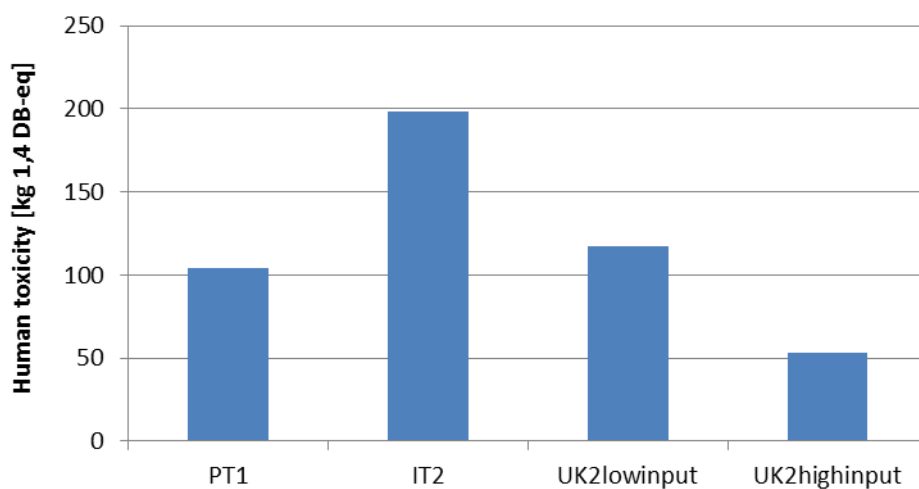


Fig. 32. Results at the agricultural stage for the Human toxicity potential. Functional Unit: 1 tonne of potatoes at the farm gate.

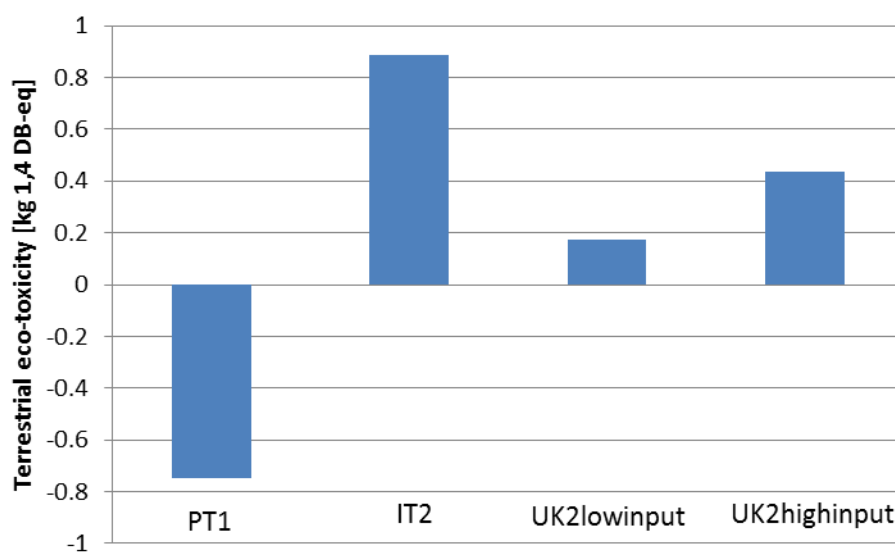


Fig. 33. Results at the agricultural stage for the Terrestrial eco-toxicity potential. Functional Unit: 1 tonne of potatoes at the farm gate.

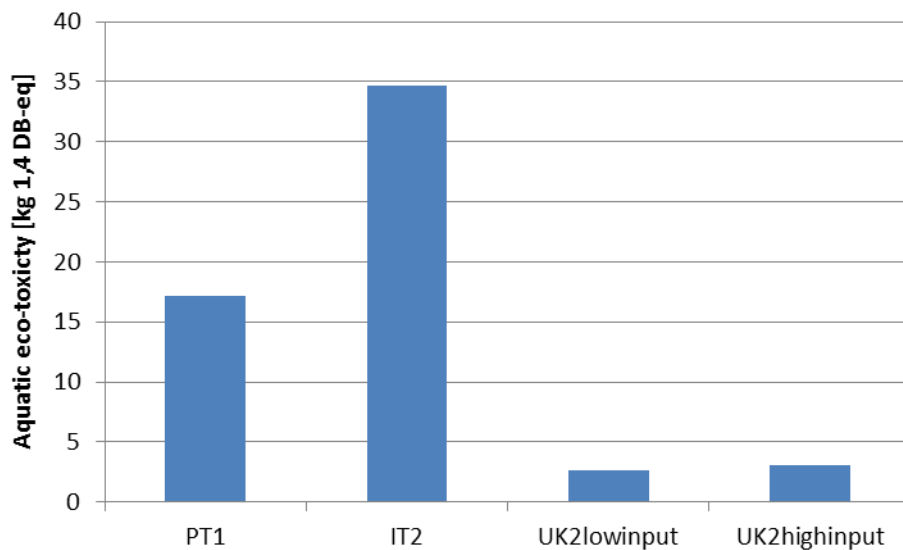
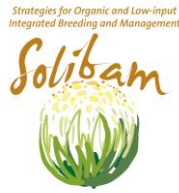


Fig. 34. Results at the agricultural stage for the Aquatic eco-toxicity potential. Functional Unit: 1 tonne of potatoes at the farm gate.

3.2.2 Results over the whole value chain - vegetables

Analysis over the whole value chain was performed for vegetable cases from the UK. Details for this study can be found in Markussen *et al.* (2014). The results are presented for 1 tonne of the vegetable mix produced by UK1 (see Table 3), to make the orders of magnitude comparable to LCA literature. The distribution phase has an important contribution to the environmental impacts of the model systems and in particular for the impact categories non-renewable energy demand (Fig. 34), global warming potential (Fig. 35) and human toxicity (Fig. 39). The use of energy in the case system was similar to the low-input system, while the impact of UK2 high had lower values. The global warming potential of UK1 was higher than both model systems. The difference in GWP between the low-input reference and the case was related to differences in management processes. The on-farm production in UK1 of seedlings and composting of woodchips, respectively, may not be as efficient as centralized production of seedlings and use of only green manure and rock phosphate for nutrient supply. The case system UK1 and the low-input reference had significantly lower aquatic eutrophication N potential (Fig. 37), and terrestrial ecotoxicity (Fig. 41) than the high-input reference. This is because these impact categories are more dependent on the applied fertilization and irrigation levels rather than on capital goods and on-farm diesel and electricity. Aquatic ecotoxicity (Fig. 41) and human toxicity (Fig. 39) effects of UK1 were also lower than both model systems. For human toxicity this was largely due to less fuel burned during distribution. For aquatic eco-toxicity the advantage was mainly at the agricultural stage and the avoidance of copper. The advantage in terms of eco-toxicity could possibly be due to the effective utilization of biological pest control; the farmer invested in preserving hedgerows, beetle banks and shelterbelts to encourage functional biodiversity. Assessment of environmental impacts exclusively from the distribution phase reveals that the local distribution system provides significantly lower environmental impacts per functional



unit for all of the impact categories considered. The relative advantage of the case system compared to the model system reached from 69% for the non-renewable energy resource use up to 98% in the case of human toxicity potential.

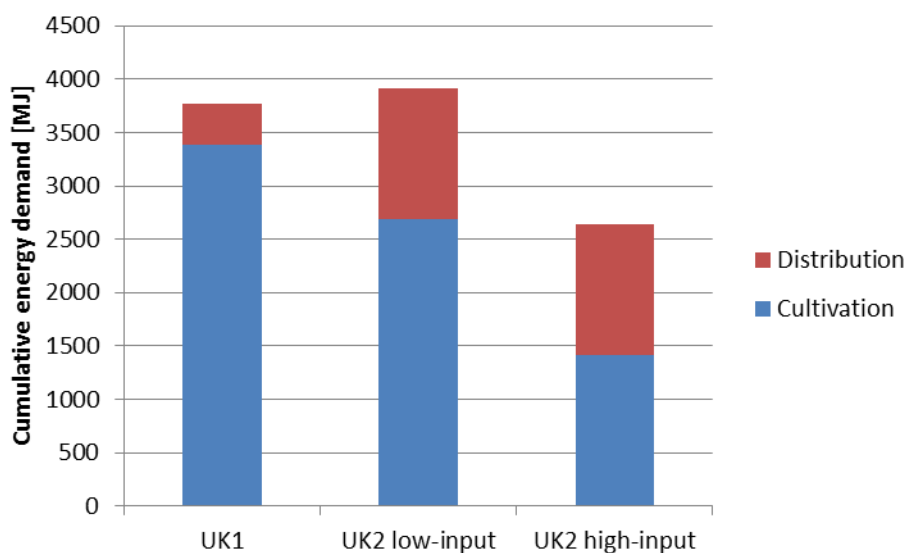


Fig. 34. Results over the whole value chain for non-renewable energy demand. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

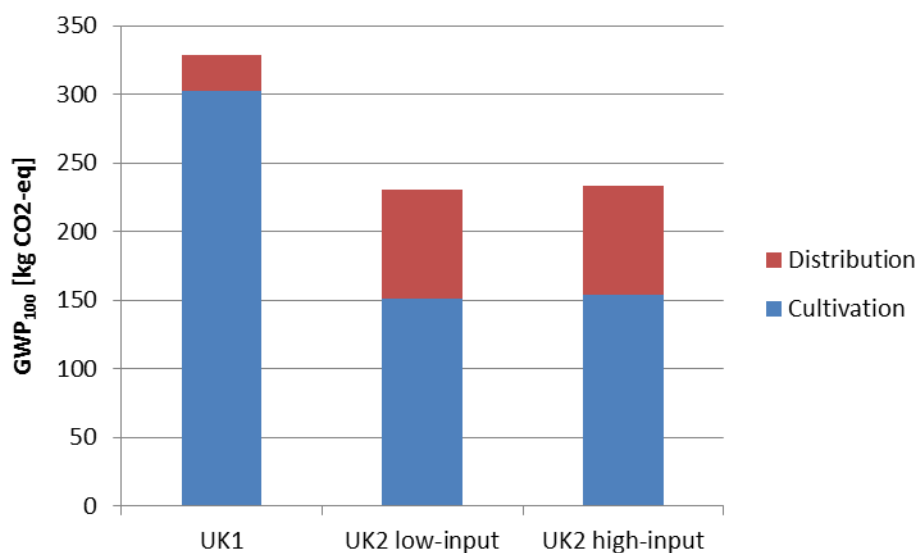


Fig. 35. Results over the whole value chain for the Global warming potential over 100 years. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

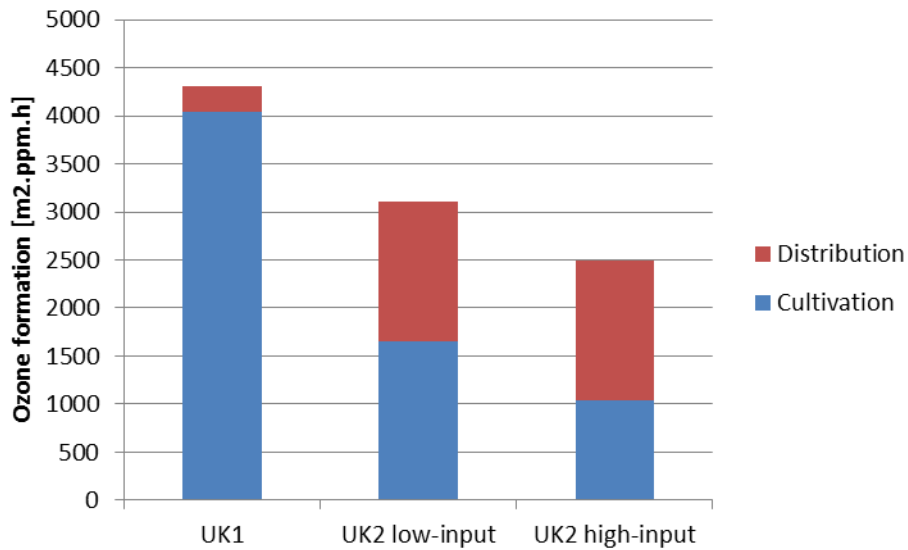


Fig. 36. Results over the whole value chain for the Ozone formation potential. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

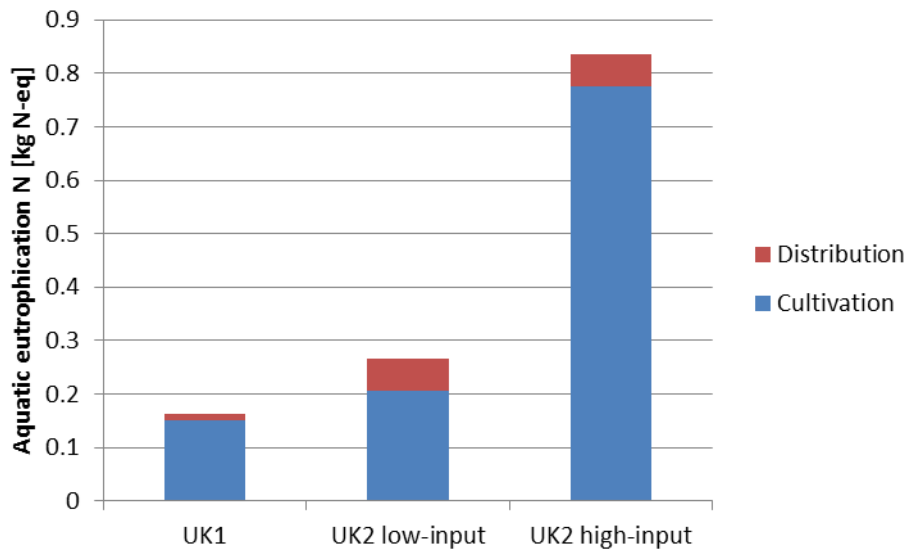


Fig. 37. Results over the whole value chain for the Aquatic eutrophication potential N. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

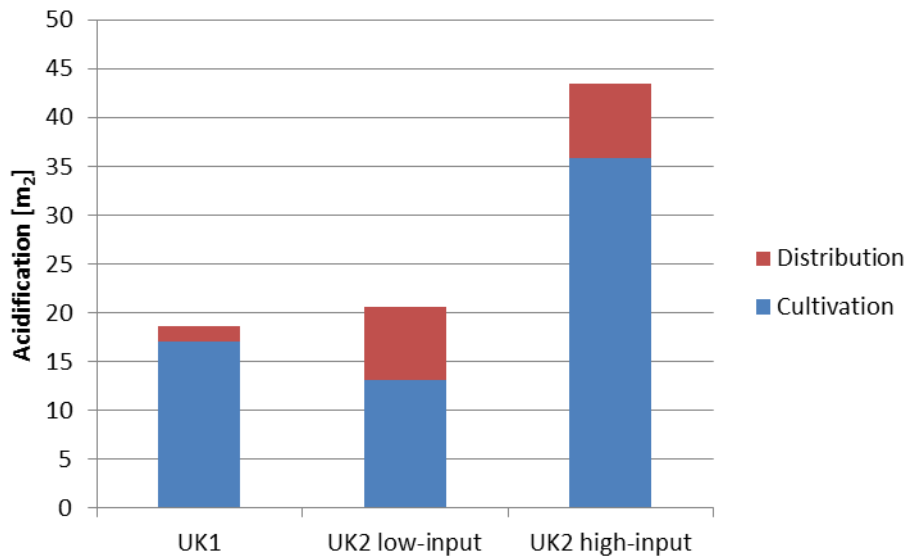


Fig. 38. Results over the whole value chain for the Acidification potential. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

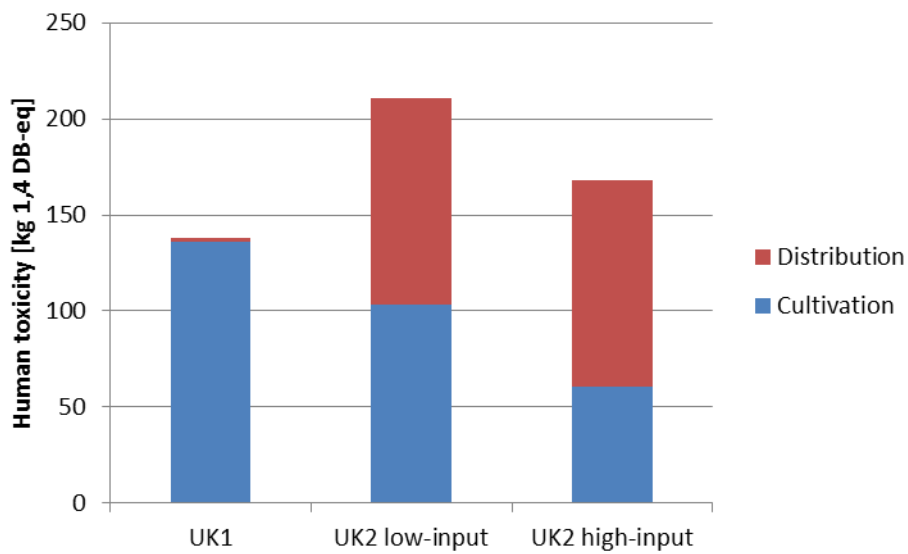


Fig. 39. Results over the whole value chain for the Human toxicity potential. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

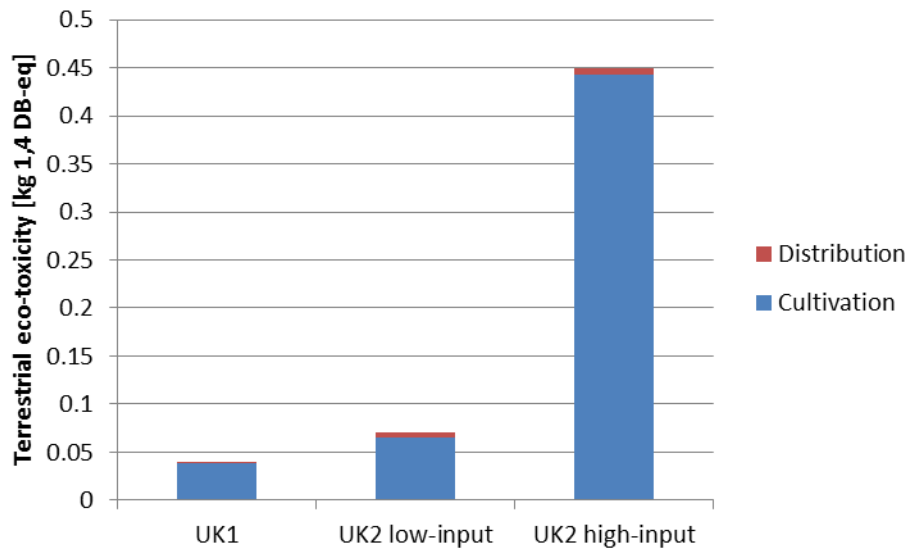


Fig. 40. Results over the whole value chain for the Terrestrial eco-toxicity potential. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

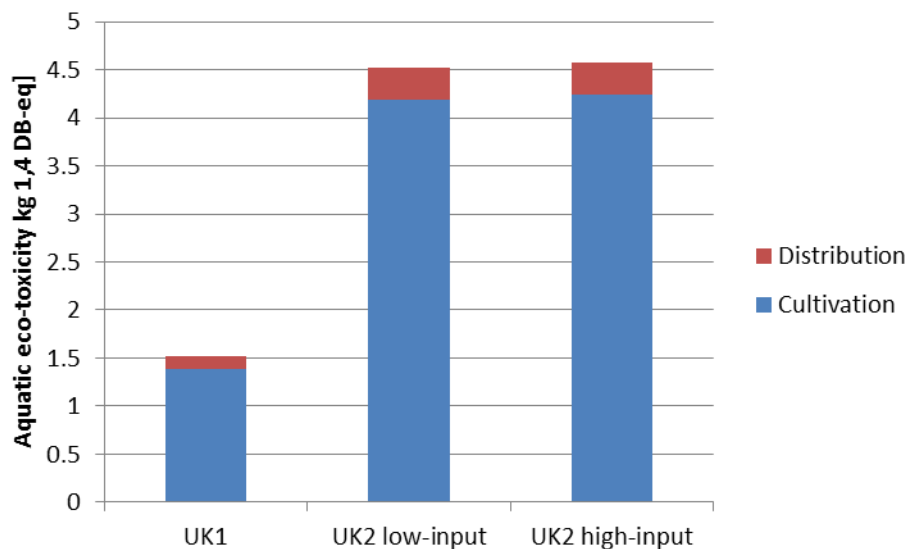


Fig. 41. Results over the whole value chain for the Aquatic eco-toxicity potential. Functional Unit: 1 tonne of mixed vegetables delivered at the consumer's door.

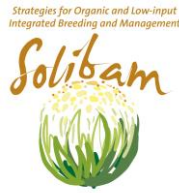
4 Results Part 2: Farm-specific improvement scenarios

Results from the interdisciplinary design workshop are presented in Table 4. Crop rotation was identified by participants as a key element to improve eco-efficiency. Eco-efficiency is defined as the ratio of environmental impacts over the amount of product. According to this definition high values mean high impacts per product unit, which is undesirable, while low values mean favourable results. Due to the low pH of soils, experts recommended soil liming



for both systems. The lists of potential management improvements were presented to the farmers who discussed their applicability with researchers during the semi-structured interview. The farmer FR1 was not willing to make significant alterations to crop rotations. According to him, long period of grassland with leguminous crops was required at his farm to produce feedstuff for his livestock, deliver nutrients for subsequent crops and improve the structure of his soil. The landrace of rye performed better in terms of grain yield than wheat at this farm with the same amount of inputs. Increasing the proportion of rye in the bread recipe could therefore reduce environmental impacts of bread. According to the farmer, the grain yield per ha could be increased two times with the introduction of drainage due to concave fields and hydromorphic soils. He expressed some concern about increased nitrate leaching potential and losses of carbon. The farmer also expressed interest in the technology of anaerobic digestion since there is an excess of farmyard manure at this farm. The second producer FR2 has agreed that increasing the proportion of rye in the bread recipe can work at this farm as well and that consumers should be able to accept darker bread as long as rye does not exceed 50% of the flour mixture. The producer expressed the opinion that due to the direct selling approach his customers can have some influence over the way their bread is made, but also he as a producer can also educate his consumers and affect their behaviour. Expanding the surface of the farm was identified as another potential solution. Emissions from capital goods constituted for a large share of environmental impacts while the farmer stated that there is a constantly increasing demand for his products so expanding the farm can also improve his economic sustainability. The mixtures that were cultivated by the farmer were characterised by very low yield potential. According to the farmer and the group of experts, there is a large potential for improving yields through breeding efforts.

Table 5 provides a description of scenarios that were considered in further LCA simulations. Predicting the yields is always associated with a high dose of uncertainty. Although the farmer FR1 stated that his yields can double as a result of drainage, we assumed a conservative 40% yield increase in this scenario. The resulting simulated yields were 2.15 t ha⁻¹ for wheat and 3.38 t ha⁻¹ for rye. To avoid nutrient depletion in the new system, we assumed 40% more manure application to the soil together with this yield increase. We have also considered the scenario of installing an anaerobic digestion plant and digesting manure instead of composting it, with subsequent digestate spread on the fields. The life cycle inventories for digestion of cattle manure were derived from the study of Poeschl *et al.* (2012). All processes related to the production of anaerobic digestion plant, its use and disposal were considered in the simulation, but only airborne emissions were considered. The methane produced in the anaerobic digestion plant was simulated to be turned into electricity, replacing some of that from the standard grid. The increase in the proportion of rye was also considered at FR2. In the increased area scenario for FR2, we have assumed the same crop rotation as currently practiced by the farmer: one third of the new land was set aside for barley and pea cultivation. The new farm design assumed 4 ha for rye cultivation, 4 ha for wheat, 3 ha of barley and pea mixture every fourth year and 2.5 ha of permanent meadow. The new farm design simulated here assumed a four years long crop rotation with 4 ha for rye cultivation, 4 ha for wheat, 3 ha for barley intercropped with pea and 2.5 ha for permanent meadow. As a result of the area increase, there was 37% more land for production of feed so we assumed less need for purchased external feedstuff in this scenario. The two horses, however, were no longer able to provide all the nutrient requirements so we assumed the need for the purchase of some external manure (under the same fertilising regime that was



currently practised by the farmer). In the improved varieties scenario, we assumed the yield potential of 2.5 t ha⁻¹ for wheat and 3.5 t ha⁻¹ for rye. According to the FAO database, wheat French farmers achieved on average 7.1 t ha⁻¹ for wheat and 5 t ha⁻¹ for rye, our estimates were therefore conservative. This means that the achievable reductions of environmental impacts are more likely to be underestimated than overestimated. Similarly to the first farm, together with the increased yield we assumed increased need for fertiliser, in this case manure, reaching the final throughput of 97 tonnes at the farm. It was estimated that following the yield increase the farm will have a surplus of 7.77 t wheat and 10.8 t rye straw per year. In the anaerobic digestion scenario, we took into account that this straw can be fed directly into the digester instead of being used as animal bedding. The previous studies showed that pure straw used as a feedstock for anaerobic digestion is more effective in terms of reducing emissions than mixed manure with straw (Poeschl *et al.*, 2012).

Table 4. Improvement proposals generated at the interdisciplinary design workshop in Rome.

<i>FR1</i>	<i>FR2</i>
Shorten the meadow time in crop rotation	Living mulch + clover with wheat
Separate permanent meadow and cash crops	Introduce landrace screening
New crops: potato, spelt, maize, fruit trees	New crops: buckwheat, hemp barley, emmer, spelt, rye
Try different legume species: red clover in the meadow, vetch as a cover crop	Buckwheat and barley breeding
Select for higher yield	Soil liming - chalk
Soil liming	

Table 5. Options considered in LCA simulations following the farmer's feedback.

<i>FR1</i>	<i>FR2</i>
Increase rye in the flour mixture (A)	Increase rye flour in the bread recipe (A)
Apply the field drainage (40% yield increase) (B)	Increase the relative farm area (B)
Anaerobic digestion of farmyard manure instead of composting (C) (no yield increase assumed)	Increase the yield by switching varieties (assumed conservative 2,5 t ha-1 for wheat, 3,5 t ha-1 for rye) (C) Anaerobic digestion of horse manure and surplus straw (D) (no yield increase assumed)

Fig. 42 and 43 show the simulated effects of management interventions on the environmental impacts of bread from two French case study farmers. The increase in the proportion of rye in the flour mixture caused simultaneous reductions of all impact categories in both case studies and field drainage caused further reductions of impacts in the case FR1. The anaerobic digestion scenario caused reductions in non-renewable energy resource use and global warming potential but increases in ozone formation and slight increase in acidification. At the second farm FR2, switching varieties caused reductions of environmental impacts for all of the impact categories considered here except for the aquatic eutrophication N. An increase in aquatic eutrophication N was caused by the need for more manure after the yield increase. The expansion of farm area caused simultaneous reduction of all environmental impacts, while anaerobic digestion caused further reduction of global warming potential at the expense of increased ozone formation. Overall, the simulated design improvements provided



reductions of all the considered environmental impacts, reaching up to 40% in the case of aquatic eutrophication potential at the farm FR1 and 60% for aquatic eco-toxicity at the farm FR2 despite conservative assumptions in scenario modelling.

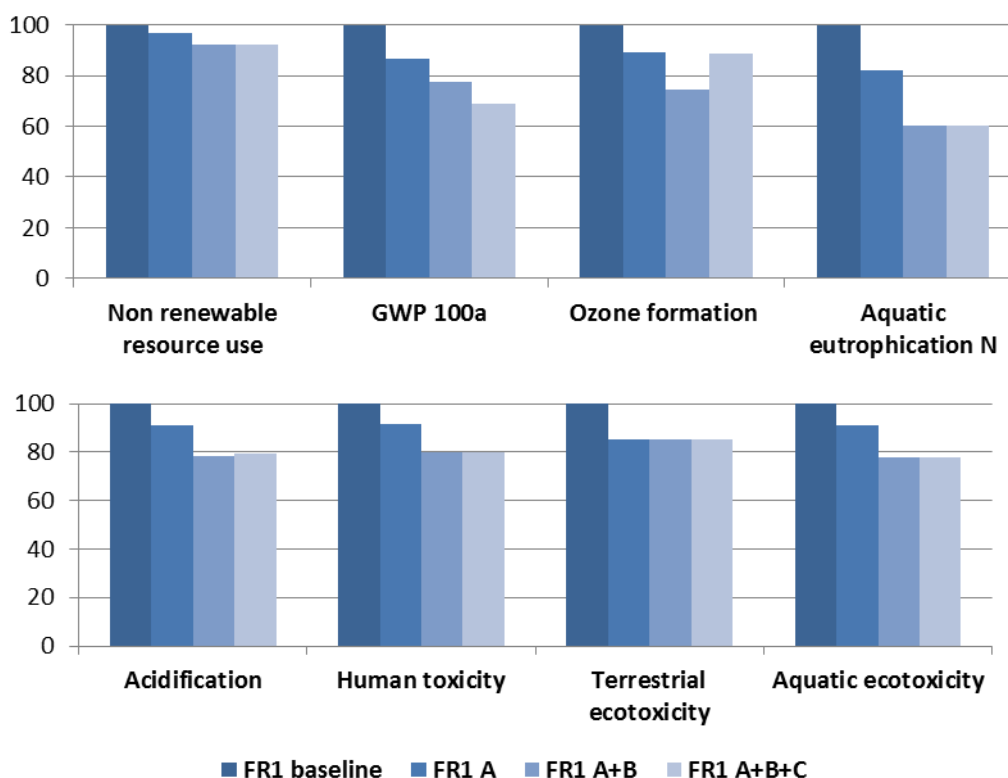


Figure 42. The effect of management improvements on the environmental impact of bread from the case FR1 (in % of the baseline). A: Increasing the proportion of rye in the bread recipe; B: Improving yields through field drainage; C: Anaerobic digestion of farmyard manure instead of composting.

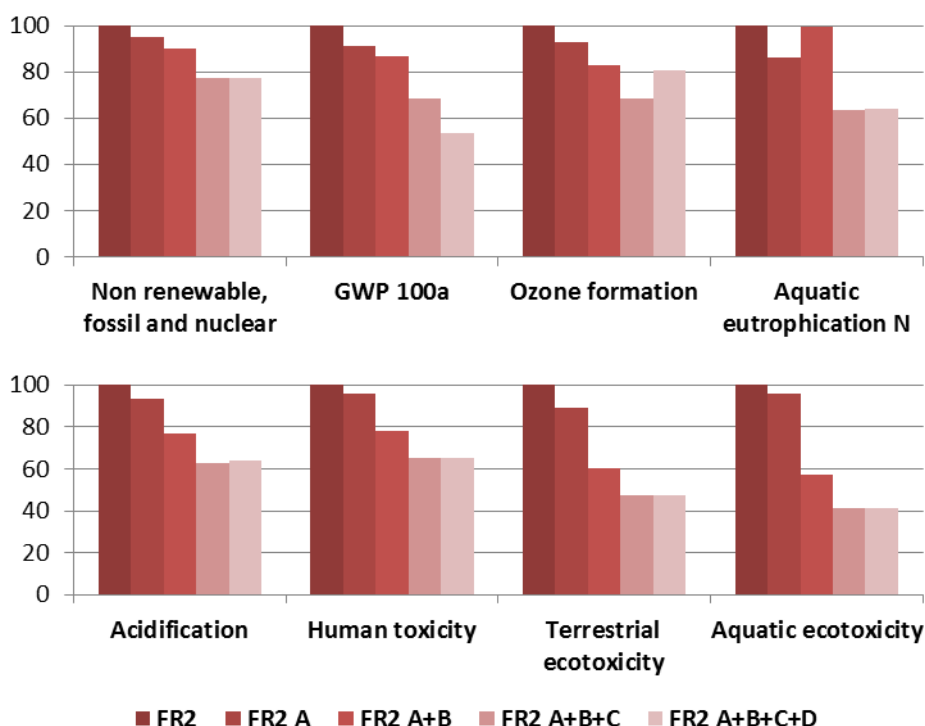


Figure 43. The effect of management improvements on the environmental impact of bread from the case FR2 (in % of the baseline). A: Increasing the proportion of rye in the bread recipe; B: Switching varieties; C: Expanding the farm area; D: Anaerobic digestion of horse manure instead of composting.

Fig. 44 shows the comparison of both system before and after the application of improvements to the generic reference-bread made of wheat from conventional farming in France, processed in an industrial bakery and distributed through the supermarket. Both farms showed slightly lower non-renewable energy resource use than the reference, especially FR2 that is using horses. For the global warming potential, FR2 had still higher impact after the improvements, although the impact was substantially reduced as compared to the baseline. The ozone formation and human toxicity of FR2 was much higher than the reference, mainly due to the large quantities of diesel used and both farms had higher aquatic eutrophication N. Acidification of the improved case FR1 was lower than the reference and both case study farms had much lower impacts on terrestrial ecotoxicity and aquatic ecotoxicity, especially after the improvements.

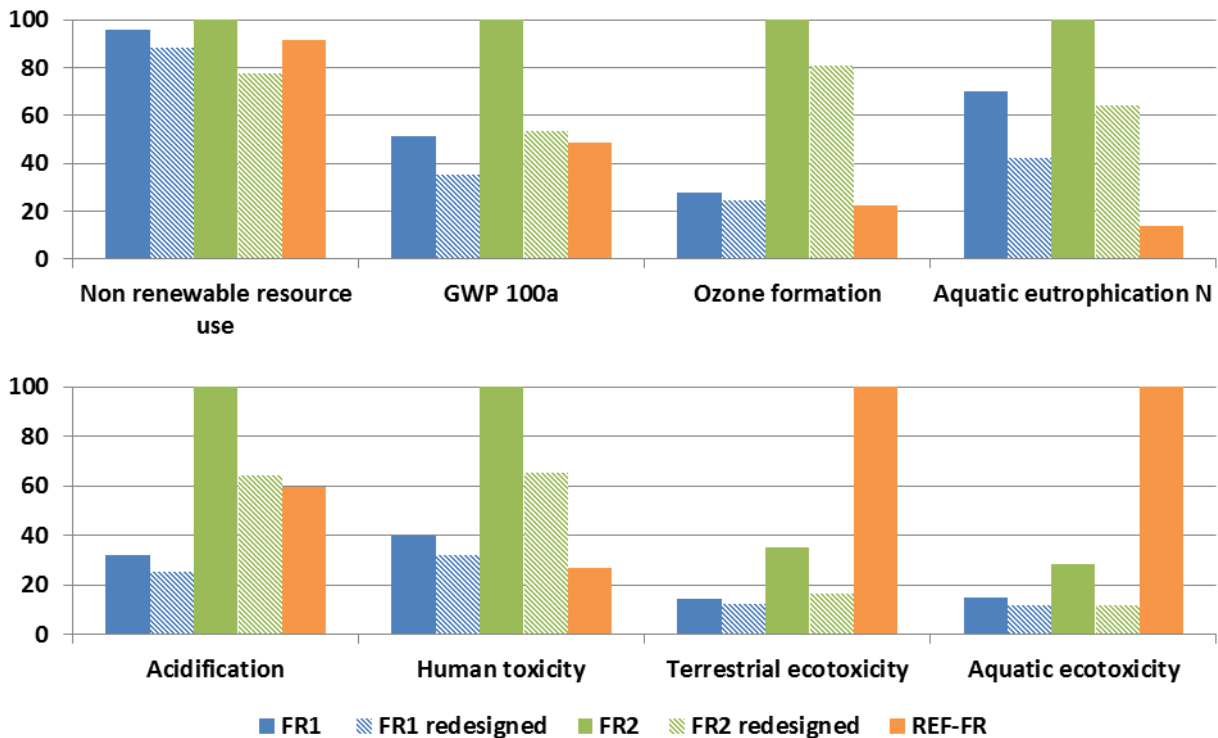
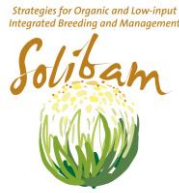


Figure 44. Relative environmental impacts of bread from the case FR1 (baseline), simulated bread from the case FR1 after the application of management improvements (FR1 redesigned), bread from the case FR2 (baseline), simulated bread from the case FR2 after the application of management improvements (FR2 redesigned) and the standard reference (REF-FR).

5 Discussion

5.1 Part 1. Environmental performance of diversified cropping systems

The detailed life cycle assessment conducted on the sample of farms in this study allows us to suggest, that diversified cropping systems have their strengths and weaknesses in terms of environmental performance. The diversified farms in the analysed sample were generally characterised by lower eco-toxicity due to the absence of pesticides and lower eutrophication than high-input organic but higher than conventional farms, although exceptions from these rules can also be found in the analysed sample. The remaining impact categories showed no consistent trend, proving to be lower in some cases and in some cases higher than the respective standard references. The evaluation at the level of the whole value chain revealed, that going beyond the boundary of the farm is necessary for the fair assessment of environmental impacts from diversified cropping systems. Due to genetic heterogeneity, products of diversified cropping systems may not always be adequate for processing in standard supply chains, while alternative supply chains organised by farmers are very different from standard supply chains and their decisions can have important environmental implications.



Low-input farmers seek to reduce the amount of farm-external inputs used and to minimise the impact of their activities on the environment as described by Parr *et al.* (1990). Lower use of purchased seeds, fertilisers and pesticides can, however, increase the need for diesel, electricity, machinery and other infrastructure to produce the same quantity of food. Van der Werf *et al.* (2007) suggested that product LCA supports intensive high-input and high-output systems that may cause local environmental problems. Our analysis demonstrated that this is not always the case, and that low-yielding systems can also be similarly or more eco-efficient than high-input ones. The wide variability of results suggests that there is scope for significant improvements in eco-efficiency within low-input agriculture.

In nearly all cases except for aquatic eutrophication potential N, results per unit of land area were higher in the reference systems than in the case study systems. This result is not surprising as analysed diversified systems were characterised by lower levels of inputs per area, mainly fertilisers, pesticides and seeds. The analysis per area gives an overview of the local impacts of farming systems. If the goal is to maintain agricultural production in a certain area, this type of analysis gives the right indicator. In general, the analysis per area unit favours extensive forms of production (Nemecek *et al.*, 2011b). Reducing input of the low-input systems further would most likely lead to lower impacts per area unit, however at the expense of lower yields. An analysis per area unit alone however is not sufficient, as it is not the only function of agriculture just to maintain a minimal form of production, but also to produce food, feed, fuel and useful materials. Therefore the productive function has also to be considered to achieve a comprehensive analysis.

As based on a small number of specific case studies, results of this analysis cannot be considered as representative for diversified low-input systems. This means that the outcomes of LCA modelling cannot be generalised to other agricultural systems. LCA studies are often based on a large number of limited datasets, generated to represent some particular type of production system (for example organic wheat production in Switzerland). Instead of generating representative life cycle inventory, this study aimed at investigating specific, real life cases with a high level of details. Although life cycle inventories based on a large number of limited datasets allow to generate representative life cycle inventories, higher level of details is required to conduct an eco-design study (Part II). This is due to the fact that opportunities for improving environmental performance are often highly site-specific. For example, the study found that the drainage of fields in the case of farmer FR1 would allow to significantly increase the yields and reduce environmental impacts. If this system was a part of a large survey of farms, such information would not be revealed and the relatively poor result would become a part of a representative sample, lowering the mean results for “average practices”. The insights gained through the analysis of the case studies in this report can be transferred to similar cases.

Besides products, agricultural systems deliver a range of other important services for the society, also called ecosystem services. These co-functions are dependent on the region of the world where the production is located. In Europe, agricultural lands have been embedded in rural landscapes for several centuries and agriculture have among others a co-function of supporting biodiversity. Many rare species of plants and animals are dependent on agricultural landscapes. Agricultural systems are also integral part of the landscape. One of the important functions of diversified cropping systems is the in-situ conservation of genetic resources. The product LCA approach that has been applied in this study does not capture some of the positive co-functions of agricultural systems. Using multiple functional units (FU) has been



the most widely used approach to multi-functionality in agricultural LCAs. In this study, the analysis at the agricultural stage was performed per product unit and per area unit. Several studies considered even more FU, adding financial approach based on the farm gross margin (Cerutti *et al.*, 2013; Nemecek *et al.*, 2011a), nutrition-based FU based on the protein content in grains (Charles *et al.*, 2006; Markussen *et al.*, 2014) or MJ of produced digestible energy (Hersener *et al.*, 2011). Consideration of multiple FU provides detailed information on the extent of environmental impacts related to each one of the analysed functions: maintaining agricultural land (area-based FU), income generation for the farmer (financial FU) or satisfaction of nutritional needs (nutrition-based FU). This makes multifunctional LCA a viable approach to provide policymakers with detailed information on all potential benefits and drawbacks of a particular farming system or technology for different stakeholder groups. Multifunctional LCA however presents some drawbacks from the eco-design perspective. One of them is the difficulty in the interpretation of results. The results of product-based LCA and area-based LCA often lead to contradictory conclusions. This bears the risk of wrong decisions, for example if weighting factors are applied to make the final choice of one solution over the other (Hayashi, 2013). Product-based LCA covers exactly the same level of inputs and outputs as the area-based LCA with the difference that in the area-based LCA the productivity of the cropping system is not factored in. This FU has therefore no relevance to eco-efficiency. None of the previously mentioned FU allows capturing all of the ecosystem services. Reducing impacts per area does not necessarily have to contribute to improving the landscape, increasing biodiversity, or providing other ecosystem services.

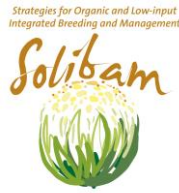
5.2 Part II. The effectiveness of integrative design

The process of integrative design with farmers and scientists participating in the scenario development and LCA used as a decision support tool allowed to achieve significant improvements of eco-efficiency in both analysed systems. This was achieved without reduction of cropping system diversity or compromising other distinctive properties, such as local production and distribution. The eco-design study of two farms allows to suggest that eco-efficiency of diversified cropping systems can have not only biophysical limitations that were mentioned in the previous paragraph, but also the lack of information on the environmental impacts of agriculture and food systems. The fact that farmers were interested in results of LCA study and that were willing to implement some of the improvement options suggest knowledge on environmental impacts of various management patterns as a factor limiting eco-efficiency.

6 Conclusions

The study revealed a high variability of environmental impacts between the farms with diversified cropping systems. This highlights the key importance of individual management decisions and suggest that there could be a significant potential for improvements of these diversified low-input systems.

In general, these diversified, low-input cropping systems been characterised by lower eco-toxicity potentials than conventional farming due to the absence of pesticides and lower eutrophication than high-input organic, due to lower inputs of organic fertilisers. However, compared to conventional farms with mainly mineral fertilisation, the eutrophication potential tend to be higher.



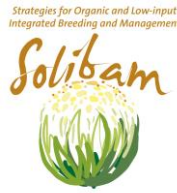
For the other impact categories, no consistent trends have been found for the diversified systems; they can have higher or lower impacts than their non-diversified counterparts. Impacts can be substantially reduced through farm-specific management improvements so that improved low-input systems can be more eco-efficient than conventional systems without compromising their secondary functions: diversity or local production.

Acknowledgements

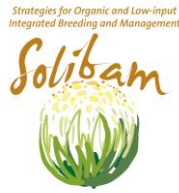
We would like to express our warm thanks to farmers that actively participated in this study. Without their collaboration, this study would not have been possible. We would also like to thank our SOLIBAM partners helping with data collection: Véronique Chable, Laurence Smith, Mads Ville Markussen, Daniela dos Santos, Carolina Passeira, Riccardo Bocci and Livia Ortolani. We thank Hanne Østergård for organising the workpackage and for many fruitful discussions within the project. This study was supported by the EU grant no. KBBE-245058-SOLIBAM.

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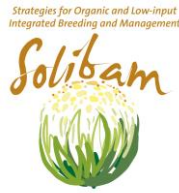
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