UNIVERSITÀ DEGLI STUDI DI MILANO

DEPARTMENT OF AGRICULTURAL AND ENVIRONMENTAL SCIENCES

Doctoral School in
"Innovazione tecnologica per le scienze agro-alimentari e ambientali"
XXVII Cycle

Improving environmental sustainability of the agro-food sector through the application of the LCA methodology

Ph.D. Thesis

Alessandra Fusi

Tutor: Prof. Riccardo Guidetti
Doctoral School Coordinator: Prof. Roberto Pretolani

Academic year 2013/2014

Contents

1. Objectives of this work

Consumers are now changing their behaviour to integrate environmental considerations into lifestyle choices. The environmental aspect is now one of the variables taken into consideration by consumers during the purchasing process. In some cases, consumers are willing to pay a premium for environmentally friendly products.

Given the importance of the environmental aspects associated with goods production, and being food consumption one of the major causes for resource use and environmental impact by modern households, the focus of this work has been the application of the LCA methodology to agro-food products in order to:

1. Depict the environmental profile of some food products, identifying the environmental hotspots associated with their production;
2. Evaluate different technical solutions in a chain optimisation perspective.

A cradle to grave perspective has been applied to some of the presented case studies (article 1 and 3) in order to identify the hotspots of the investigated products along their whole life cycles. This approach allows evaluating phases, such as packaging and transport, which might contribute considerably on the environmental impact of a product.

A cradle to farm-gate perspective has been applied to three case studies (article 2, 4 and 5): being the agricultural phase one of the most critical stage of cereals production, the attention was therefore focused on that step of the life cycle when cereals (maize, wheat, triticale and rice) were assessed. Finally a gate to gate approach has been used in article 6: being cooking one the most impactful phase of pasta, this stage of the life cycle was the object of the analysis.

With reference to the above mentioned goals of LCA application, article 4, 5 and 6 could be framed as a chain optimisation application (Point 2), while the first two are aimed at defining the environmental profile and hotspots of a product (Point 1). Article 3 is in between the application of LCA as "environmental profiling" and "chain optimisation". The classification of these articles based on such LCA application categories is presented in Figure 1.

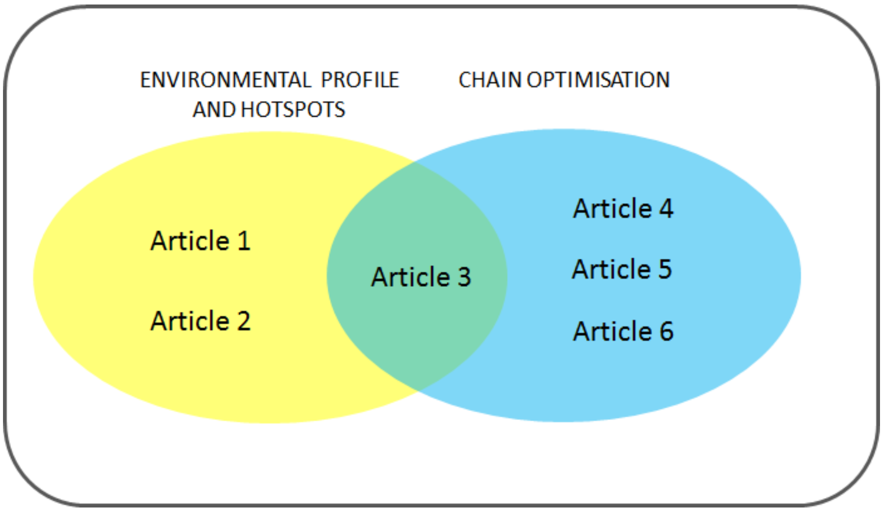


Figure 1. Presented articles and their classification.

2. What development means and what it is supposed to achieve

The notion of human development incorporates all aspects of individuals' well-being, from their health status to their economic and political freedom. According to the *Human Development Report 1996,* published by the United Nations Development Program, "human development is the end, economic growth a means". Economic growth, obtained by increasing a nation's total wealth, also enhances its potential for reducing poverty and solving other social problems. But history offers a number of examples where economic growth was not followed by similar progress in human development. Instead growth was achieved at the cost of greater inequality, higher unemployment, weakened democracy, loss of cultural identity, or overconsumption of natural resources needed by future generations.

The recent decades show that there is no automatic link between growth and human development. More attention must go to the structure and quality of that growth in order to ensure that it is directed to sustain human development, reducing poverty, protecting the environment and ensuring sustainability.

If the trade-offs between the economic benefits (i.e. additional incomes earned by the majority of the population) and the environmental and social/human losses resulting from economic growth are unbalanced, the overall result for people's wellbeing becomes negative. Thus such economic growth becomes difficult to sustain politically. Moreover, economic growth itself inevitably depends on its natural and social/human conditions. To be sustainable, it must rely on a certain amount of natural resources and services provided by nature, such as pollution absorption and resource regeneration. Economic growth must be constantly sustained by the products of human development, such as higher qualified workers along with opportunities for their efficient use: more and better jobs, better conditions for new businesses to develop, and greater democracy (Figure 2).

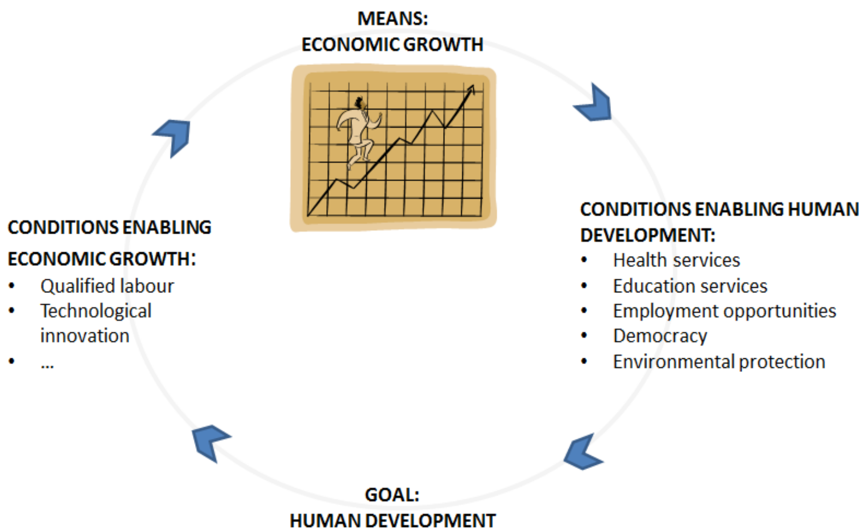


Figure 2. Economic growth and human development (Source: T. Soubbotina, 2004).

3. Sustainable development

The word sustainability is derived from the Latin *sustinere* which means “maintain", "support", or "endure”. The first use in the modern sense of the term sustainable was done by the Club of Rome in 1972 in its report on the *Limits to Growth*, written by a group of scientists. Describing the desirable *state of global equilibrium*, the authors used the word *sustainable*: "We are searching for a model output that represents a world system that is sustainable without sudden and uncontrolled collapse and capable of satisfying the basic material requirements of all of its people".

The United Nations World Commission on Environment and Development (WCED) in its 1987 report *Our Common Future* gave what would become the most popular definition of sustainable development: "Development that meets the needs of the present without compromising the ability of future generations to meet their own needs".

On 8 September 2000, at the headquarters of the United Nations, the General Assembly adopted the *Millennium Declaration* which identifies principles on sustainable development, including economic development, social development and environmental protection. Within this document the term *sustainable development* relates to three different domain: economics, environment and social sustainability. The 2005 World Summit on Social Development confirmed the importance of all these three domains when referring to sustainable development. The economic, social and environment components were later defined as the three pillars of sustainability (Figure 3).

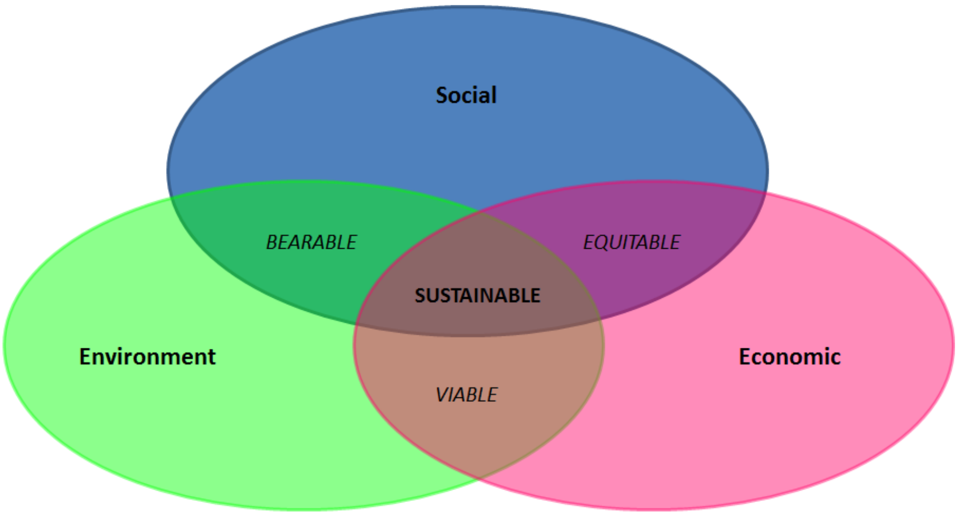


Figure 3. The three pillars of sustainability.

In more detail, the social sustainability includes:

- Human health: protect, sustain, and improve human health;
- Resource security: protect, maintain, and restore access to basic resources (e.g. food, land, and energy);
- Democracy and governance: provide democratic processes;
- Quality of life: ensure that basic needs are met;
- Equity: provide equitable opportunities and outcomes for all members of the community, particularly the poorest and most vulnerable ones.

Economic sustainability means:

- Jobs: create or maintain current and future jobs;
- Incentives: generate incentives to encourage sustainable practices;
- Natural Resource Accounting: incorporate natural capital depreciation and ecosystem services in cost benefit analysis.

Environmental sustainability consists of:

- Air quality: attain and maintain air-quality standards and reduce the risk from toxic air pollutant;
- Water quality: reduce exposure to contaminants in drinking water and in recreational waters;
- Stressors: reduce effects by stressors (e.g. pollutants, greenhouse gas emissions) to the ecosystem;
- Resource Integrity: reduce waste generation, increase recycling, and ensure proper waste management; restore resources by mitigating and cleaning up accidental or intentional releases;
- Ecosystem Services: protect, sustain, and restore the health of critical natural habitats and ecosystems;
- Green Engineering & Chemistry: develop chemical products and processes to reduce/prevent chemical hazards, reuse or recycle chemicals, treat chemicals to render them less hazardous, dispose of chemicals properly.

4. How to measure sustainability

In a context in which sustainability becomes, or should become, part of the development process, it is essential to measure such sustainability in order to understand whether a product/process/activity is sustainable or not. To accomplish that goal, different tools have been developed. The main methodologies available today are:

- Life Cycle Assessment (LCA), which evaluates the interactions between the environment and the product or an activity, considering the entire life cycle of the product/activity under evaluation;
- Emergetic Analysis, which allows to determine the amount of solar radiation required to obtain a product or a flow of energy for a given process;
- The Embodied Energy Analysis, which enables to convert all inputs used in the production of a product in an amount of oil equivalent;
- Carbon Footprint, which quantifies the greenhouse gas emissions associated with the life cycle of a product. At the international level, guidelines have been established in order to define a common calculation method (PAS 2050, GHG Protocol);
- Water Footprint, which quantifies the volume of potable water used (and polluted) to produce a good;

- Ecological Footprint, which measures the area of biologically productive land and sea needed to regenerate the resources consumed by a certain activity and to absorb the corresponding waste.

Among the above-mentioned methods, the last three are a subset of the results of an LCA study, as they take into consideration the effects of the life cycle of a product in relation to a single environmental parameter.

4.1 Life Cycle Assessment: a first definition

Among the methods cited before, the LCA methodology has gradually assumed a prominent role. Such method has been defined as "a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impact of those energy and materials used and releases to the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing, extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling, and final disposal" (SETAC, Society of Environmental Toxicology and Chemistry) (Figure 4).

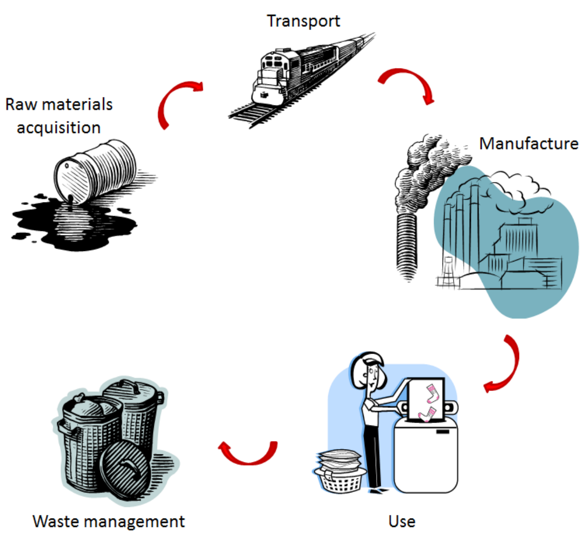


Figure 4. Life cycle of a generic product.

4.1.1 History of LCA

Life cycle assessment origins go back to 1960s when concerns over the limitations of raw materials and energy resources led to the development of a method which enabled the quantification of resources use. However interest in environmental studies decreased in the late 1970s, to be on the rise again in the late 1980s when the growth of environmental awareness refocused the attention on LCA as a potential and valuable environmental management tool. The

first international meetings for LCA practitioners and researchers were held in 1990 and 1991 under the patronage of the Society for Environmental Toxicology and Chemistry (SETAC).
The importance of LCA was recognised by the International Standards Organisation (ISO) in the late 1990s, when a series of four ISO LCA standards were published. During these years, the interest in the use of LCA as a decision-supporting tool has increased and a new topic in the LCA literature emerged. The latter was called "Life Cycle Management (LCM)" and it was defined as follows (Remmen et al., 2007):
"[..] a product management system aiming to minimise environmental and socio-economic burdens associated with an organisation's product or product portfolio during its entire life cycle and value chain. LCM is making life-cycle thinking and product sustainability operational for business through the continuous improvements of product system […]".
As easily deducible from the above definition, LCM recognises the key role played by the use of a life-cycle perspective in a product-oriented environmental management.
Over the last decade, further validation of LCA methodology as appropriate support tool for environmental management initiatives has stemmed from the great interest in carbon footprint and water footprint. The UK's PAS 2050 (BSI, 2008) is in fact largely based on LCA methodology as established in the ISO standards.
As for the application of LCA to the agro-food sector, in the mid-2000s great interest was put in the development of LCA-oriented tools and assessments. In 2007, the UK retailer Tesco launched the initiative of labelling all its product with carbon footprint information and other retailers such as Marks & Spencer and Walmart announced their commitment to reduce the carbon footprint throughout the supply chains of products sold in their outlets.

4.1.2 The four steps of an LCA study

According to the ISO LCA standards, an LCA analysis should be structured in four separate phases:

1. Goal and Scope Definition: goal and scope are to be based in relation to the intended application.
2. Inventory Analysis (LCI): this step involves the actual collection of data. The output of this phase is a table which quantifies all relevant inputs and outputs of the product system.
3. Impact Assessment (LCIA): during this phase the results of the Inventory Analysis are translated into environmental impacts (e.g. climate change, eutrophication, acidification, ozone depletion).
4. Interpretation: at this stage, conclusions and recommendations for decision-makers are drawn.

Figure 5 represents these phase and their iterations.

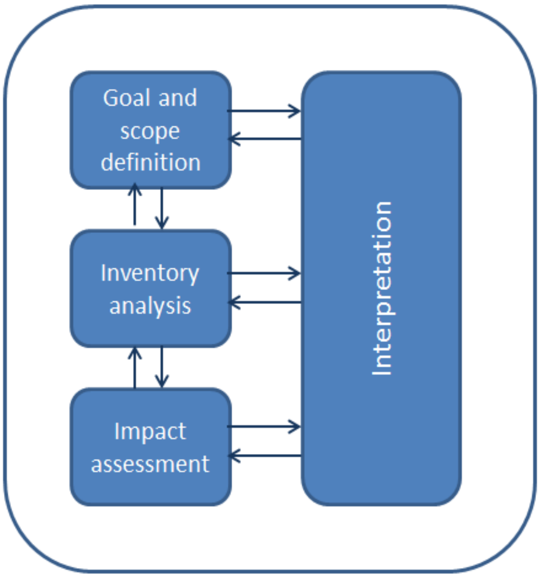


Figure 5. The phases of LCA (Source: ISO, 2006)

In the following section the above mentioned LCA stages will be discussed in detail.

1.1.2.1 Phase 1

Guidelines for the first phase of an LCA study are presented in the ISO 14041:2006. This stage of an LCA analysis involves the definition of the following aspects:

1. Goal of the study;
2. Scope of the analysis.

Goal of the study

During the goal definition the intended applications of the study are identified as well as the targeted audience. The goal definition is decisive for all the other phases of the LCA:

- The goal definition guides to set the frame for the LCI and LCIA stages;
- The final results of the LCA are interpreted in accordance with the goal of the study: a clear, initial goal definition is hence essential for a correct interpretation of the results.

Six aspects that should be addressed during the goal definition:

1. Intended application(s): The following LCA applications are the most frequently used ones:

    - Identification of Key Environmental Performance Indicators (KEPI) of a product;
    - Comparison of specific goods or services;
    - Benchmarking of specific products against the product group's average;
    - Development of Product Category Rules (PCR) or a similar specific guide for a product group;

- Development of a life cycle based environmental declaration (e.g. Environmental Product Declaration (EPD));
- Greening the supply chain.

2. Limitations due to the method and assumptions.
3. Reasons for carrying out the study, namely the drivers and motivations, and especially identify the decision-context. Different decision-context situations can be differentiated:
   - Studies on decisions: case of a study that is to be used to support a decision;
   - Studies of descriptive character: case of a study that does not imply a direct decision-support.
4. Target audience of the study, i.e. to whom the results of the study are intended to be communicated. Typically the target audiences can be identified as: "internal" vs. "external" and "technical" vs. "non-technical". Different types of target audiences imply different scoping requirements on documentation.
5. Comparative studies to be disclosed to the public: it should be explicitly stated whether the LCA study includes a comparative assertion intended to be disclosed to the public.
6. Commissioner of the study and other influential actors.

Scope definition

The scope definition implies the description of the following factors:

1. The system or process that is studied and its function and the functional unit.
   System under study:
   It is recommended to provide a detailed description of the analysed system (product or activity), whose function(s) should be precisely defined.
   As shown in Figure 6, the product system can be seen as part of a wider "Background system" which is in turn part of the "environment". While the "Foreground system" includes the economic processes which directly contribute to the product system, the "Background system" involves all the economic processes that contribute to the "Foreground system" such as material and energy production. The "Environment" is the place where human economic activities take place.
   In other words, the foreground processes are those that are under direct control of the producer of the good, the background system includes those processes that are not under direct control or decisive influence of the producer of the good. The background processes and systems are hence outside the direct influence or choice of the producer.
   The system boundaries define which parts of the life cycle and which processes belong to the analysed system. They separate the system under study from the rest of the technosphere. At the same time, the system boundaries also define the boundary between the analysed system and the ecosphere[1] (Figure 7).
   There are different approaches that may be taken into account when defining the system boundaries (Figure 8):

[1] In the ISO 14044:2006 the term ecosphere is referred to as *"environment".*

- Gate to gate: it is the simplest option and includes the analysis from reception of the raw materials to the end of production, when the product is ready to be used or received by the final user without considering distribution;
- Cradle to gate: this option includes some additional life cycle stages such as the extraction of raw materials and their transportation;
- Cradle to grave: this approach goes one step further and includes, within the scope of the analysis, the distribution, the use and the end of life management of the product.

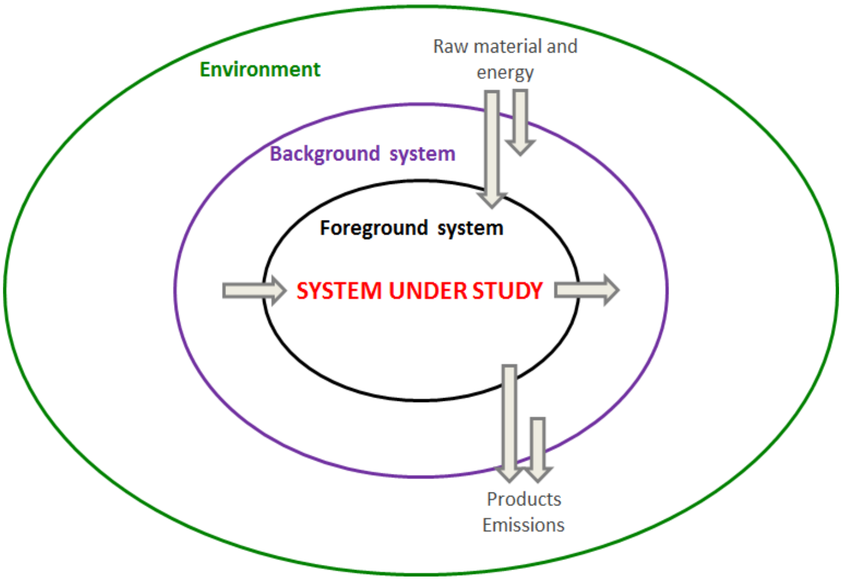


Figure 6. Foreground and Background systems (Source: Cowell, 1998).

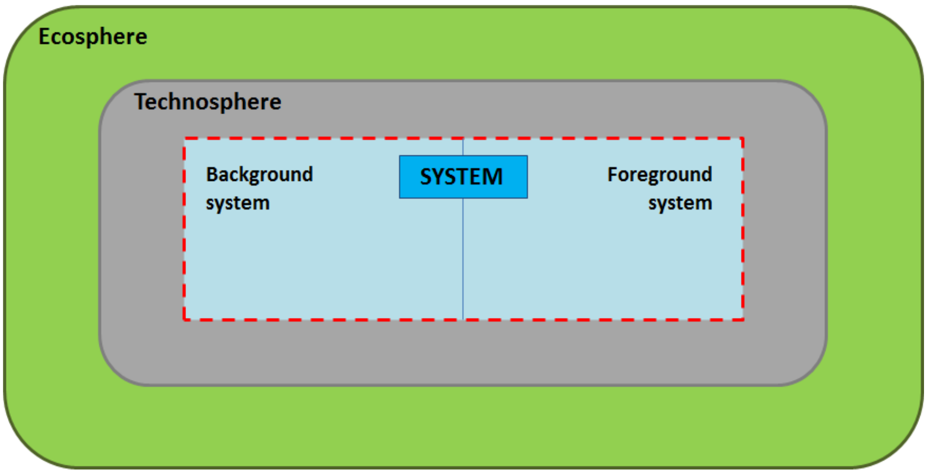


Figure 7. The analysed system's boundaries (dashed border), separating it from the remainder of the technosphere and from the ecosphere (Source: ILCD handbook, 2010).

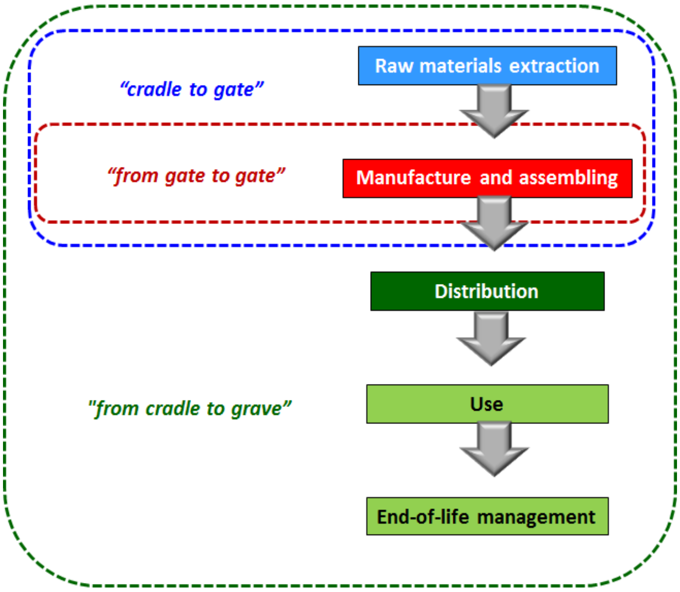


Figure 8. Cradle to grave, cradle to gate and gate to gate approaches.

<u>Functional unit:</u>
The functional unit represents the reference unit to which relate all the inputs and outputs of the system and with which express the results of the analysis.
The functional unit quantifies the qualitative and quantitative aspects of the function(s) of the system under investigation along the questions "what", "how much", "how well", and "for how long".
The functional unit shall be identified and the following aspects should be taken into consideration:

- function provided (what);
- in which quantity (how much);
- for what duration (how long), applicable to durable goods;
- to what quality (in what way and how well is the function provided).

The qualitative definition of the system's function(s) is a description of the way in which the function(s) are provided and of other qualities of the product.
In many cases more than a single product (and therefore, more than a single function) can result from one process. When such condition occurs, the process is called multifunctional (Figure 9).

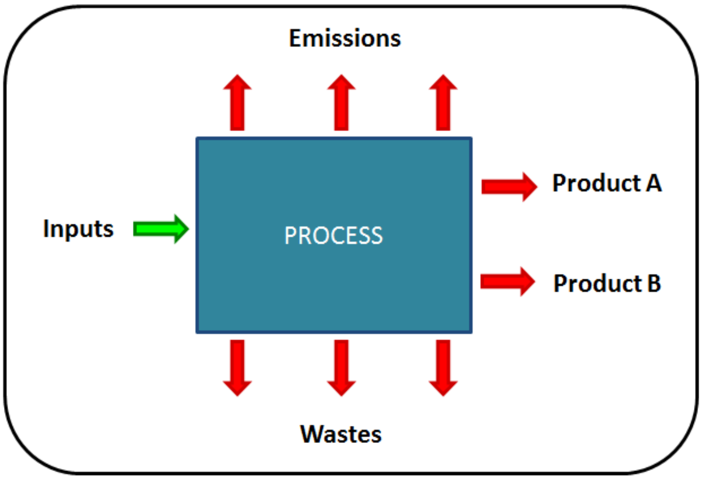


Figure 9. Generic multifunctional process (Source: ILCD Handbook, 2010).

2. System boundaries, completeness requirements, and related cut-off rules;
3. LCIA impact categories to be covered and selection of specific LCIA methods to be applied;
4. LCI data quality requirements regarding technological, geographical and time-related; representativeness and appropriateness;
5. Types, quality and sources of required data and information.

1.1.2.2 Phase 2 (Life Cycle Inventory)

During this phase the actual data collection and modelling of the system has to be carried out. Such activities implicate time and resources, making the LCI phase the most effort requiring step of an LCA study.

In addition, along this phase, decisions about the following aspects have to be made:

1. Modelling principles: attributional or consequential modelling;
2. Approaches to solve multifunctionality: allocation or system expansion / substitution approaches

Modelling principles

The attributional life cycle inventory modelling principle is referred to as “descriptive”: it depicts the potential environmental impacts that can be attributed to a system (e.g. a product) over its life cycle. Attributional modelling makes use of historical, fact-based, measureable data of known uncertainty, and includes all the processes that are identified to relevantly contribute to the system. In attributional modelling the system is hence modelled as it is or was.

The consequential life cycle model depicts the generic supply-chain as it is theoretically expected in consequence of the analysed decision. The consequential life cycle inventory modelling principle is called “change-oriented”, “effect-oriented”, “decision-based”. It aims at identifying the consequences that a decision in the foreground system has for other processes and systems of the economy. The consequential life cycle model is hence not reflecting the actual (or forecasted) specific or average supply-chain, but a hypothetic generic supply-chain is modelled.

Solving multifunctionality

If a process provides more than one function, i.e. delivering several goods and/or services (often also named simplified "co-products"), it is multifunctional.
The problem about multifunctional processes is that in LCA a single system is to be analysed in order to determine the specific environmental impact which can be related to its life cycle. The ISO 14044:2006 presents a hierarchy of different approaches to this multifunctionality problem:

1. Subdivision (preferable approach): the multifunctional process under investigation contains physically distinguishable sub-process steps and it is theoretically possible to collect data exclusively for those sub-processes (Figure 10).
2. System expansion: adding for the given case missing functions.
3. Substitution: defining an “avoided” process with subsequent “avoided” impacts, expanding the system boundaries and substituting the not required function with an alternative way of providing it. It means to subtract the inventory of another system from the analysed system (
4. Figure 11). This often leads to negative inventory flows and it can even result in negative overall environmental impacts for the analysed system.
5. Allocation: partitioning the input and/or output flows of a process to the product system under study on the basis of:
   - physical relationship, such as mass and energy content;
   - economic value.

If possible, according to ISO 14044:2006, allocation should be performed in accordance with the underlying causal physical relationship between the different products. In practice there is often the difficulty to clearly identify the most appropriate allocation key.

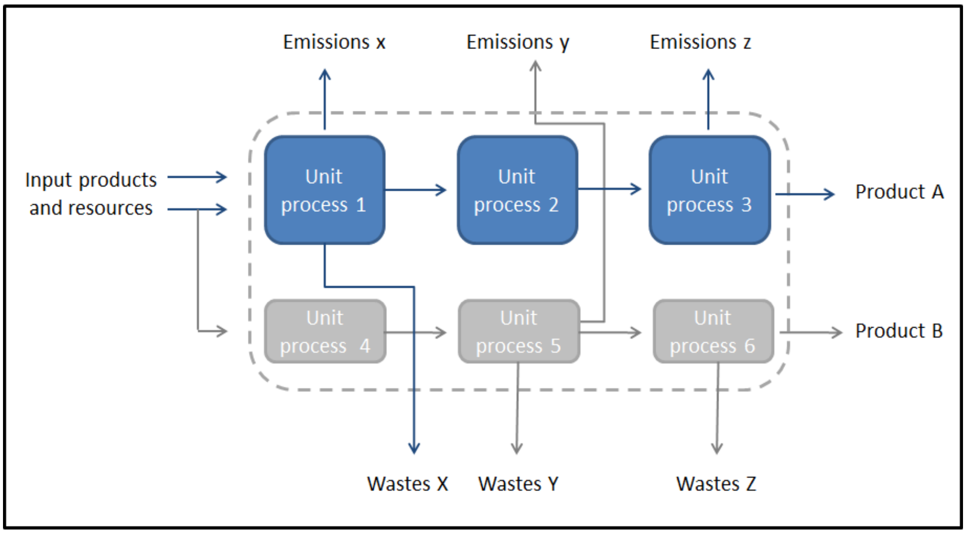


Figure 10. Subdivision approach for solving multifunctionality (Source: ILCD Handbook, 2010).

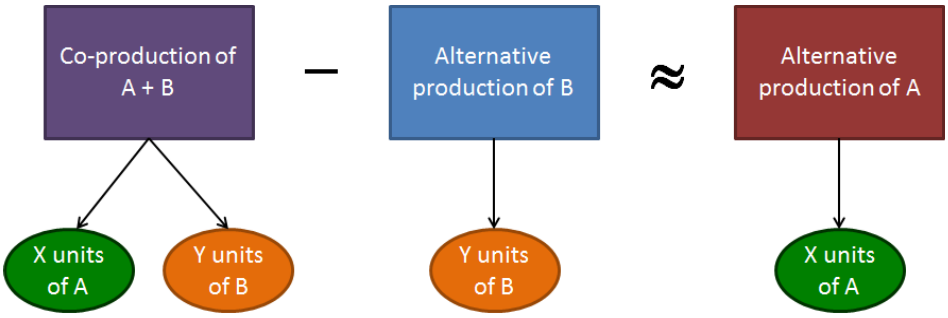


Figure 11. Solving the multifunctionality problem by substitution (Source: ILCD Handbook, 2010).

Collection of data

Setting the system boundaries means deciding which life cycle stages, activities, specific processes, and elementary flows to include and which to omit from the life cycle model. The system boundaries should be represented in a graphic diagram displaying the life cycle stages of the system that are initially intended to be included in the analysis. Such activity is functional to the collection of data which is to be carried out during this phase.

In general, all processes and flows that are attributable to the analysed system are to be included in the system boundaries. However, not all these processes and elementary flows are quantitatively relevant. If not relevant, data of lower quality can be used or entirely cut-off.

The requirements in terms of the system data collection regard:

- raw materials;
- products;
- solid waste and emissions to the environment.

Two categories of data can be distinguished:

1. Primary data: obtained through direct measurements, interviews or annual reports;
2. Secondary data: procured from generic sources such as LCA databases, literature and previous LCA studies.

For the foreground system, specific inventory data (primary data) should be used, while for the background system, data can be sourced from available background databases (secondary data). It is nonetheless important that all foreground and background data used are methodologically consistent and that the quality requirements for the analysed system are met.

Cut-off criteria

Systems flows which are not relevant from a quantitative point of view can be ignored: they are cut-off.

Cut-offs are quantified in relation to the percentage of environmental impacts that is estimated to be excluded through the cut-off.

Data Quality

Quality control of the collected data is an important part of data collection. This activity includes:

- identifying significant issues ;
- completeness check;
- sensitivity check;
- consistency check.

As for data quality, the following parameters should be taken into account:

- Time-related coverage;
- Geographical coverage;
- Technology coverage;
- Precision, completeness and representativeness of the data;
- Consistency and reproducibility of the methods used throughout the data collection;
- Uncertainty of the information and data gaps.

1.1.2.3 Phase 3 (Life Cycle Impact Assessment)

In the Life Cycle Impact Assessment (LCIA) phase, emissions and resource data identified during the LCI are translated into indicators that reflect environment pressures as well as resource scarcity. This calculation is based on factors (generally calculated using models) which represent the predicted contribution to an impact per unit emission or resource consumption. Several LCIA methodologies have been developed since the 1990s; the existence of a wide range of LCIA methods makes it difficult to compare LCA results.
This phase comprises the following sub-phase:

3. Selection and Definition of Impact Categories: identifying relevant environmental impact categories (e.g. global warming, acidification, eutrophication).
4. Classification: assigning LCI results to the impact categories (e.g. classifying carbon dioxide emissions to global warming).
5. Characterisation (Figure 12): calculating the relative contributions of the emissions and resource consumption to each impact category (e.g. all greenhouse gas emissions are aggregated into one indicator for global warming). Such calculation is based on scientific models.
   Impact characterisation uses conversion factors, called characterisation factors, to convert the LCI results into representative indicators of impacts. Impact indicators are typically obtained using the following equation:

$$Impact\ Indicators = Inventory\ Data\ x\ Characterisation\ Factor$$

6. Normalisation: expressing potential impacts in ways that can be compared. Normalised LCIA results are obtained by dividing the LCIA results by the normalisation basis, separately for each impact category.
7. Weighting: emphasising the most important potential impacts. Weighing methods are based on values and preferences regarding environmental issues expressed by society. In weighting, the typically initially normalised LCIA results for the different impact categories are multiplied with a relative weighting factor.

The ISO 14044, which regulates this stage of LCA studies, states that he first three steps (impact category selection, classification, and characterization) are mandatory.
The selection of impact categories must cover all relevant environmental issues related to the analysed system. This is unless in the goal definition a limitation was set as e.g. in case of Carbon footprint studies, where exclusively Climate change is considered.
Table 1 lists the most commonly used impact categories.

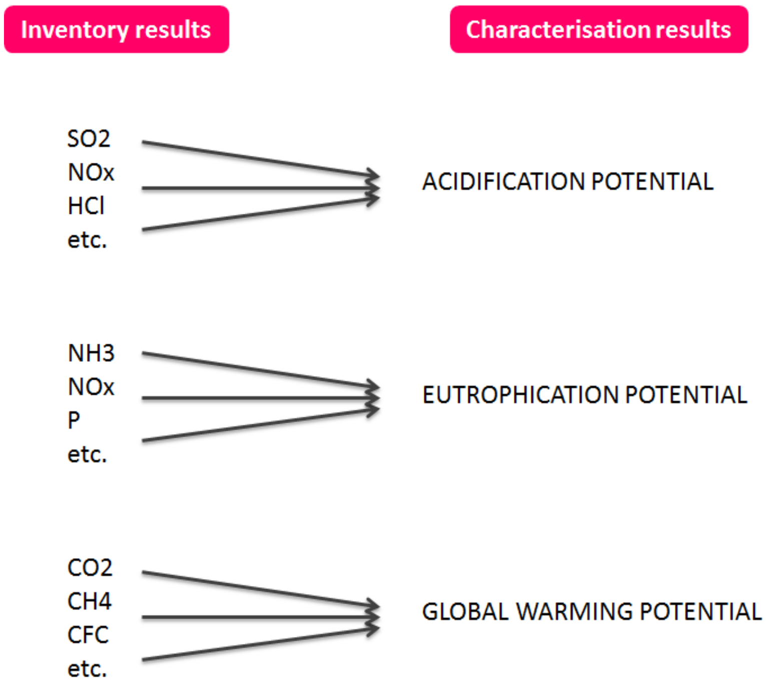


Figure 12. The characterisation step (Source: Sonesson et al., 2010).

Table 1. Most commonly used impact categories (Source: EPA, 2006).

Impact category	Scale	Examples of LCI data	Common characterisation factor	Description of characterisation factor
Global warming	Global	Carbon dioxide Nitrogen dioxide Methane Chlorofluorocarbons Hydrochlorofluorocarbons	Global Warming Potential	Converts LCI data to carbon dioxide (CO_2) equivalents

Acidification	Regional Local	Sulfur Oxides Nitrogen Oxides Ammonia	Acidification Potential	Converts LCI data to hydrogen (H+) ion equivalents
Eutrophication	Local	Phosphate Nitrogen Oxide Nitrogen Dioxide Nitrates Ammonia	Eutrophication Potential	Converts LCI data to phosphate (PO4) equivalents
Stratospheric ozone depletion	Global	Chlorofluorocarbons Hydrochlorofluorocarbons Halons	Ozone Depleting Potential	Converts LCI data to trichlorofluoromethane (CFC-11) equivalents
Photochemical smog	Local	Non-methane hydrocarbon	Photochemical Oxident Creation Potential	Converts LCI data to ethane (C_2H_6) equivalents
Terrestrial toxicity	Local	Toxic chemicals with a reported lethal concentration to rodents	LC_{50}[a]	Converts LC_{50} data to Equivalents
Aquatic toxicity	Local	Toxic chemicals with a reported lethal concentration to fish	LC_{50}[a]	Converts LC_{50} data to Equivalents
Human health	Global Regional Local	Total releases to air, water, and soil	LC_{50}[a]	Converts LC_{50} data to equivalents
Resource depletion	Global Regional Local	Quantity of minerals and fossil fuels used	Resource Depletion Potential	Converts LCI data to a ratio of quantity of resource used versus quantity of resource left in reserve
Land use	Global Regional Local	Quantity disposed of in a landfill and land modifications	Land Availability	Converts mass of solid waste into volume using an estimated density
Water use	Regional Local	Water used or consumed	Water Shortage Potential	Converts LCI data to a ratio of quantity of water used versus quantity of resource left in reserve

[a] The average concentration of a chemical or mixture in air as a gas, vapour, mist, fume or dust capable of killing 1/2 of the test animals exposed by inhalation under specific conditions.

As was mentioned before, normalisation is an optional step under ISO 14044:2006. Normalised LCIA results give, for each impact category, the relative share of the impact of the analysed system on the total impact of this category per average citizen or globally. When displaying the normalised LCIA results of the different impact topics, it can be seen to which impact topics the analysed system contributes relatively more, and such results can support the interpretation of the impact profile

Weighting is, as well, an optional step under ISO. It involves assigning distinct quantitative weights to all impact categories expressing their relative importance. As the normalisation step, weighting

represent a support tool towards the interpretation of the results. Also to implement the cut-off criteria, the use of weighted and normalised LCIA results is used.
The normalised and weighted LCIA results can also be summed up across all impact categories, providing a single score which summarises the overall impact of the analysed system.

Impact categories – some explanations

Climate change, eutrophication and acidification are ranked high on the policy agenda and the role of food production towards the above-mentioned impacts is relevant. Given their importance, a detail description of these impact categories will be provided below.

Climate change

The average global temperature has increased by 0.74°C during the last 100 years, causing a number of changes such as the rise of global average sea levels, more intense and longer droughts or the decline of snow cover and mountain glaciers in both the hemispheres.
Carbon dioxide from fossil fuel combustion represents the least important Green House Gas (GHG) emitted from the food sector, while biogenic methane and nitrous oxide are the major contributor to food's carbon footprint. Emissions of CH_4 and N_2O in agriculture were responsible for 10-12% of world global emissions in 2005 (IPCC, 2007). According to FAO the global livestock production alone makes up 18% of the total GHG emissions.
The atmospheric concentration of methane has increased by 150% over the last centuries, while the rise of atmospheric N_2O is around 18%. Table 2 lists the anthropogenic sources of methane and relatives emission amounts.

Table 2. Anthropogenic sources of methane and relatives emission amounts (Source: Denman et al., 2007).

Anthropogenic source	**Emission (Tg CH_4/y)**
Energy production	82-104
Landfill and organic waste	35-69
Ruminants	76-92
Rice cultivation	31-112
Biomass burning	14-88
Total	*238-465*

The contribute to climate change of each GHG emission is calculated through its Global Warming Potential (GWP) index, which is a relative measure of how much heat a greenhouse gas traps in the atmosphere. It compares the amount of heat trapped by a certain mass of the gas in question over a given period of time (usually 100 years) to the amount of heat trapped by the same quantity of carbon dioxide over the same time horizon. GWP indexes are based upon radiative properties of GHG, namely: the radiative efficiency (infrared-absorbing ability) of each gas relative to that of carbon dioxide, as well as the decay rate of each gas (the amount removed from the atmosphere over a given number of years) relative to that of carbon dioxide.
A definition of the GWP has been provided by the IPCC (1990) as the ratio of the time-integrated radiative forcing from the instantaneous release of 1 kg of a trace substance relative to that of 1 kg of a reference gas:

$$\frac{\int_0^{TH} a_x \cdot [x(t)]\, dt}{\int_0^{TH} a_r \cdot [r(t)] dt} = GWP(x)$$

where TH is the time horizon over which the calculation is considered, a_x is the radiative efficiency due to a unit increase in atmospheric abundance of the substance in question (i.e., Wm^{-2} kg^{-1}), [x(t)] is the time-dependent decay of the instantaneous release of the substance. The corresponding quantities for the reference gas (carbon dioxide) are in the denominator.
Table 3 shows lifetimes and radiative efficiencies for carbon dioxide, methane and nitrous oxide.

Table 3. Lifetimes and radiative efficiencies of CO_2, CH_4 and N_2O (Source: IPCC, 2007).

GHG	Lifetime (years)	RadiativeEfficiency (W m^{-2} ppb^{-1})
CO_2	200-500 [a]	$1.4x10^{-5}$
CH_4	12	$3.7x10^{-4}$
N_2O	114	$3.03x10^{-3}$

[a] Carbon dioxide has a variable atmospheric lifetime. Unlike other greenhouse gases in fact , carbon dioxide does not undergo a simple decline over a single predictable time scale.

Aquatic eutrophication

Aquatic eutrophication is defined as nutrient enrichment of the aquatic environment. Excess of nutrients increases the production of fast growing algae and because of such phenomenon, the water becomes turbid. Nutrient enrichment, if driven to a far extent, determines anaerobic or low-oxygen conditions and results in significant mortality of fish resources (Figure 13).
Increased human interference of the nitrogen and phosphorous cycle during the 20^{th} century is to be considered the principal cause of the eutrophication problem. A significant rise of nitrogen fluxes in rivers have been recorded in Europe and the US. Phosphorous load has increased as well.
Emissions of nitrogen to water from agriculture occur mainly as nitrate leaching (the magnitude of which depends on farming systems, soil types and climate) or through discharged effluents from manure waste storages.
As for phosphorous, its release from agricultural activities is due to soil erosion, surface runoff and leaching.

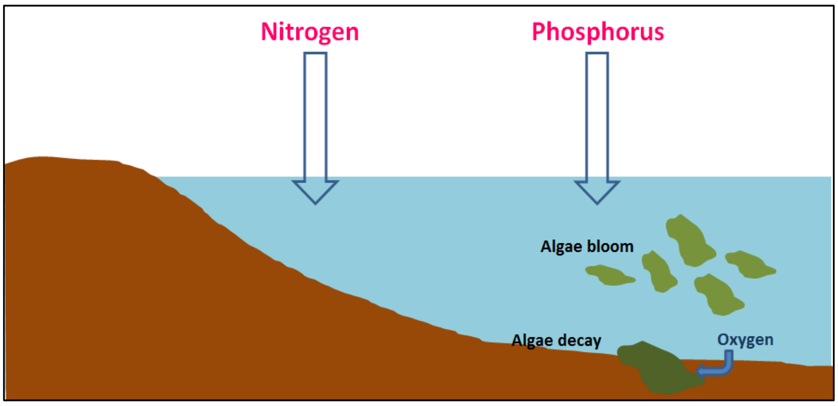


Figure 13. Aquatic eutrophication.

Terrestrial eutrophication

Terrestrial eutrophication includes the effects of excess nutrients on plant functioning and species composition in terrestrial ecosystems. Vegetation in natural resources is mainly influenced by the limited availability of nitrogen. Human activities condition atmospheric N deposition, the increase of which is responsible of changes in ecosystems. The emissions of ammonia, largely derived from animal production and, to some extent, from application of synthetic N-fertilisers, strongly influence the deposition of ammonia to soil, causing damages both at local and regional level.

Acidification

Acidification can lead to different environmental damages such as reduced forest and plant health, loss of aquatic life, leaching of toxic aluminium. The major acidifying substances are nitrogen oxides (NO_x), sulphur (SO_2) and ammonia (NH_3). The latter represents the acidifying compound of major importance in food production . Its acidifying effect is the result of the following chemical process, through which ammonia is converted into nitrate:

$NH_4^+ + 2O_2 \rightarrow NO_3^- + H2O + 2H^+$

Midpoints vs endpoints

LCIA methods exist for midpoint and for endpoint level.
The LCIA mid-point approach is also known as "problem-oriented" approach or classical impact assessment method. The term mid-point refers to the category indicator for each impact category which is expressed in the mid pathway of impact between LCI results and end-point. The end-point LCIA methodology is also known as "damage-oriented" approach. Figure 14 and Figure 15 schematically represent the midpoint and endpoint approaches.
On midpoint level a higher number of impact categories is differentiated and the results are more accurate and precise compared to the category endpoint. Below, the most commonly used impact categories at midpoint level and the category endpoints:

- Impact categories: Climate change, Ozone depletion, Human toxicity, Photochemical ozone formation, Acidification (land and water), Eutrophication (land and water), Ecotoxicity, Land use, Resource depletion (minerals, fossil and renewable energy resources, water).
- Category endpoints: Damage to human health, damage to ecosystem, depletion of natural resources.

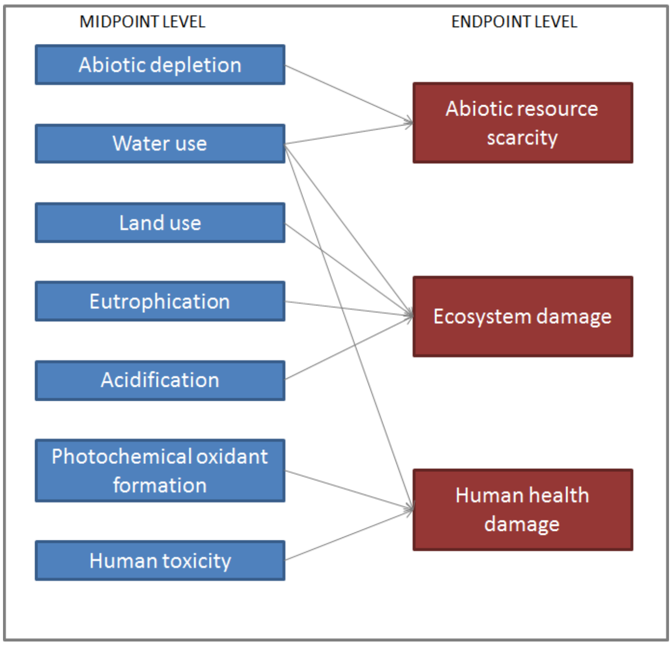


Figure 14. Midpoints and related endpoints for some impact categories (ILCD handbook, 2010)

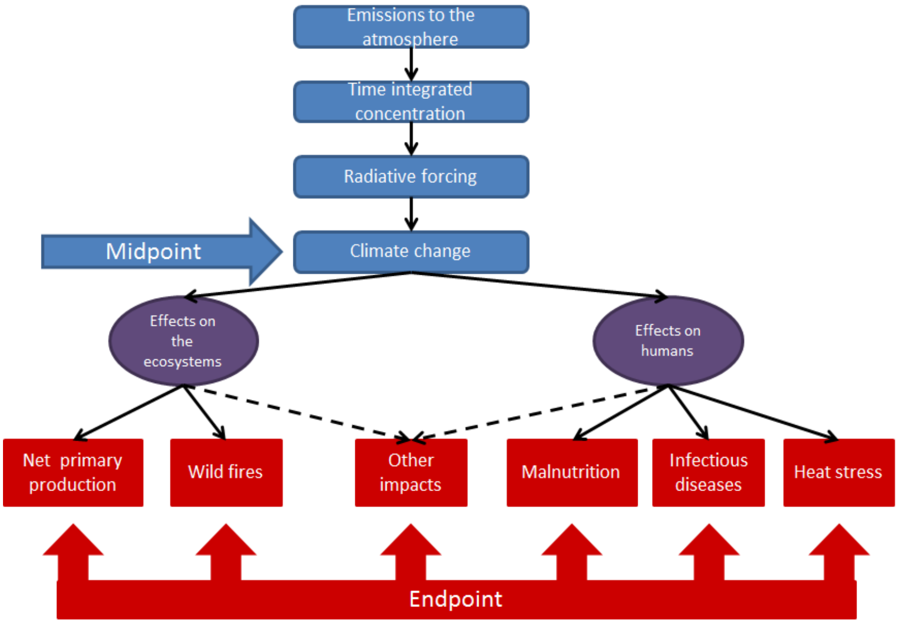


Figure 15. Climate change (midpoint) and its effects (endpoints) (Source: ILCD handbook, 2011)

1.1.2.4 Phase 4

The Interpretation phase of an LCA has the main purpose of deriving robust conclusions and recommendations. The results of the life cycle assessment are evaluated in order to answer the

questions posed in the goal definition. The interpretation is related to the intended applications of the LCA study and is used to suggest recommendations.
Conclusions and recommendations have to be consistent with the intentions and restrictions of the goal and scope definition of the LCA study. This especially relates to the appropriateness of the functional unit and the system boundaries, as well as the achieved data quality, in relation to the goal. The interpretation should present the results of the LCA in an understandable way and help the stakeholders of the LCA study to understand the robustness of the conclusions and any potential limitations of the LCA study.
The interpretation phase consists of three activities, as schematically illustrated in Figure 16 and detailed below:

- The significant issues (i.e. the key processes, parameters, assumptions) are identified. The purpose of this first element of interpretation is to analyse and structure the results of previous phases of the LCA study in order to identify the significant issues, such as:
    - the main contributors to the LCIA results, i.e. the most relevant life cycle stages, processes and elementary flows, and the most relevant impact categories
    - the main choices that have the potential to influence the precision of the final results of the LCA (methodological choices, assumptions, foreground and background data used in the inventory, LCIA methods, normalisation and weighting factors used, if applied).
- These issues are evaluated with regard to their sensitivity or influence on the overall results of the LCA. Such evaluation involves:
    - Completeness check
    - Sensitivity check
    - Consistency check

  The outcome of this evaluation is crucial to give strength to the conclusions and recommendations of the study and it is hence important that it is presented in a clear and understandable way for the stakeholders.
- The results of the evaluation are used in the formulation of conclusions and recommendations from the LCA study.

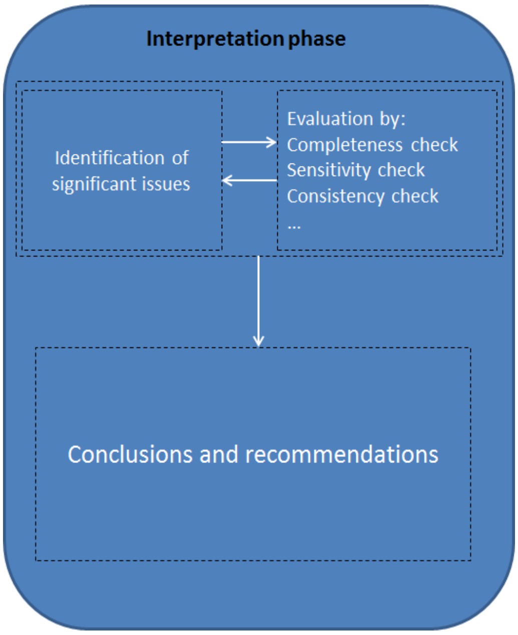


Figure 16. The interpretation phase (Source: ILCD handbook, 2010).

Completeness check

Completeness checks are performed on the inventory in order to determine the degree to which it is complete and whether the cut-off criteria have been met.
Completeness of the inventory, in relation to the initially defined cut-off criteria, relates to:

- Process coverage: coverage of all relevant processes in the system;
- Elementary flow coverage: coverage of all relevant elementary flows in the inventories for the processes of the system which appear to be relevant for the impact categories considered;
- Additional relevance criteria for elementary and waste flows: also those emissions and wastes should be include in the data collection that have a low mass and energy content but a known relevance for the respective type of processes or industry;.
- Leaving out negligible flows: it is optional to leave out negligible flows that jointly make up less than 10 % of the share of impact that is cut-off.

Sensitivity check

The interpretation phase assesses the influence on results of variations in process data choices and other variables. In the sensitivity analysis, these changes are deliberately introduced in order to gauge the robustness of the results.
Accordingly with ISO 14043, the sensitivity and uncertainty analysis should be based on those model choices known to be of major influence on the results of the study, such as:

- allocation rules;
- boundary setting;
- process data;

- cut-off criteria;
- characterisation method: alternative characterisation methods, which could be adopted instead of the baseline method ;
- normalisation data and weighting method (if carried out).

Once one or more variables of the list above are selected, the changes produced by their variation in the LCA results should be analysed.

Consistency check

The consistency check is carried out to investigate the consistency of:

- Methods and assumptions: methodological issues of relevance are especially the LCI modelling frameworks (i.e. attributional or consequential) and approaches (i.e. allocation criteria and selection of substituted systems), but also setting of system boundaries, extrapolations of data, the consistent application of the impact assessment (i.e. consistent application of the LCIA elements, including, if used, normalisation and weighting factors).
- Data: inventory data issues of relevance concern the consistency of the time-related, geographical, and technological representativeness of the data, the appropriateness of the LCI results to represent processes in the foreground and background system, and the completeness and precision of the data.

4.1.3 Application of LCA: a decision-making instrument, a searching tool for potential improvements &a communication instrument

Decision-making tool

Product development has been considered as the main field of application of the LCA since the origin of this methodology. Product development is seen in fact as decisive for achieving sustainability in industrial societies, making the life-cycle perspective an indispensable approach.
Product development is a complex process consisting of several phases, such as planning, conceptual design, detailed design. It is realised with the contribution of interdisciplinary teams and characterised by the constant need of trade-offs between performance issues (performance, shelf-life, aesthetics)and production costs, market constraints and legal requirements. Environmental aspects need to be included along this process.
The available literature on eco-design highlights the importance of considering the environmental issues at the early stage of the product development process, when environmental concerns may have great influence on the product design. However, at this stage, when no concrete design yet exists and few data are available, carrying out an LCA study is pretty challenging. That is why several simplified LCA approaches have been developed for product development purposes. Such simplified methods are however often based on full-scale LCA studies.
Besides product development, also process development represents an application area for LCA (Figure 17 and Figure 18). Process development focuses, as the name says, on the process by which products are produced or disposed, rather than the products themselves. The LCA perspective could support decisions on the most suitable process configuration, from an

environmental point of view. As a matter of fact, policy drivers for cleaner production increasingly take a life-cycle perspective.
In order to have a more detailed and complete idea of the process to be developed, LCA is often merged with other modelling techniques such as process simulation.
Another field of application of the LCA technique is “green purchasing”, which is dependent on environmental information about products. LCA is one of the possible information tools in supporting green purchasing.
Moreover, life-cycle approaches are central to several environmental policies and regulations. Examples from the European context include the directive on eco-design and policies for green public procurement . The importance of the LCA approach in supporting sustainability, is also evidenced, at the European level, by *The Sustainable Consumption and Production Action Plan*, which includes a series of proposals on sustainable consumption and production.

Searching tool for potential improvements

Two of the most widespread LCA applications are the identification of “hot-spots” and the search for potential improvements. These applications, which may lead to decision making, are aimed at getting to know the environmental strength and weaknesses of a product from a life-cycle point of view.

Communication tool

Marketing played a central role in the development of LCA and in its standardisation at the ISO level.
LCA was and it is used as methodological base for eco-labelling, environmental declarations and carbon footprinting. Eco-labels can be seen as a “seal of approval” for environmentally benign products and can therefore be attractive for marketing purposes. Ecolabels at the same time convey information to the consumer in a simple and objective way, enabling individuals to include environmental concerns in their own decisions along with considerations on price and quality. The EU ecolabelling scheme has so far resulted in criteria for 12 product groups:

- Washing machines
- Soil improvers
- Kitchen towels
- Laundry detergents
- T-shirts and bed linen
- Paints and varnishes
- Dishwashers
- Toilet paper
- Double-ended light bulbs
- Single ended light bulbs
- Copying paper
- Refrigerators

Environmental product declarations (EPD) are verified documents that report environmental data of products based on life cycle assessment and other relevant information and in accordance with the international standard ISO 14025. EPD are based on Product Category Rules (PCR), which enable to perform LCA studies and related environmental declarations in a consistent and comparable way. PCR are defined as a set of requirements and guidelines specific for homogeneous groups of products.

Furthermore, always for marketing purposes, LCA is used in many other ways, such as communicating the fact that life-cycle approaches are used or simply communicating life-cycle information in a non-standardised format.

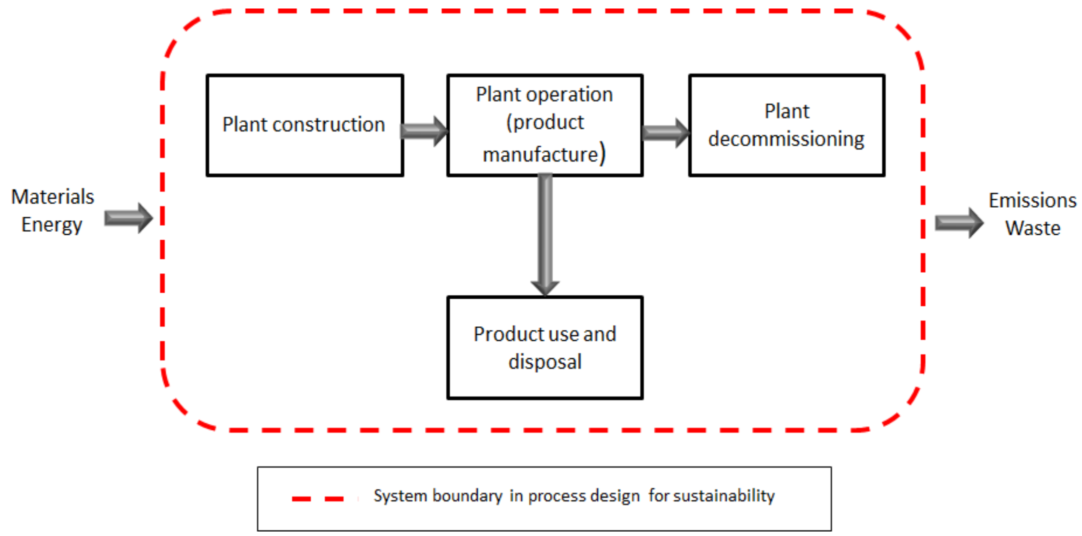


Figure 17. Process design for sustainability (Source: Azapagic et al., 2006).

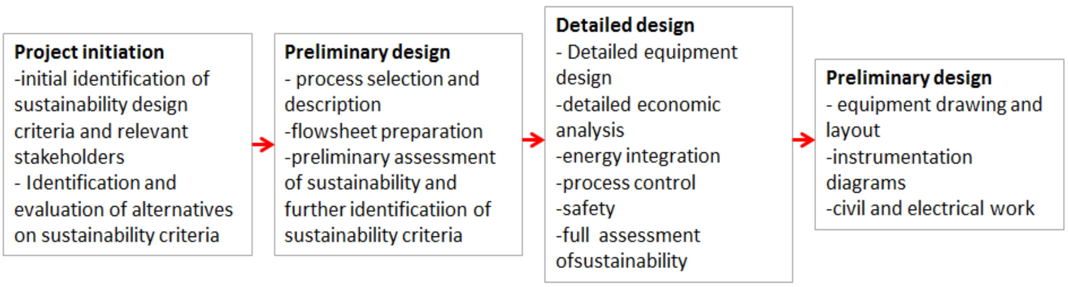


Figure 18. Stages in process design for sustainability (Source: Azapagic et al., 2006).

4.1.4 Limitations of the LCA methodology

A multitude of different methodological choices (allocations systems, system boundaries, characterisation methods, etc.) can be made when carrying out an LCA study. This, on the one hand, allows the LCA to be a flexible tool and, on the other hand, makes it difficult to compare different studies having as their object the same category of product. Therefore, (further) standardisation of LCA after ISO is required in order to solve lack of results comparability.

Another weakness of the LCA methodology is that the results have a low spatial and temporal resolution, and that social and economic aspects are not taken into account. Being based on steady-state and linear-homogeneous modelling, it is very difficult to include in the LCA model spatial and temporal characteristics and nonlinear characteristics of large numbers of processes that occur all over the world.

Moreover, notwithstanding the fact that LCA takes into account only the environmental dimension of sustainability, its results are difficult to communicate. This is due to the complexity of the method which entails, at the same time, another limitation: the need of high expertise to conduct an LCA analysis. Together with high expertise, a life-cycle study requires an extensive examination and it is therefore time consuming and expensive.

4.1.5 Future trends of LCA

Given the importance of the LCA methodology and not forgetting about its embodied limitations, it is reasonable to assume the following development trends of such tool:

1. Standardisation of life-cycle-based assessments methods for different product categories. Even though the ISO standards provide generic framework for carrying out LCA analysis, more specific guidelines are needed in order to make it possible to compare alternative products within a single product category. These rules of thumbs should instruct on the following issues: inclusion or exclusion of particular unit processes, allocation method, definition of functional unit.
2. Standardisation of life-cycle-based assessment methods for different impact categories. Recent success of carbon footprint and water footprint has resulted in international initiatives aimed at their standardisation: the ISO 14067 and the ISO 14046 on carbon and water footprinting, respectively.
3. Communication of LCA results, where the following key issues concerning environmental labelling have to be disentangled:

    - Whether absolute values for environmental impacts or values relative to the performance of a reference product should be displayed;
    - Information on whether impacts have been reduced over a defined time frame;
    - Whether one single weighted value or several values representing different impact categories should be displayed.

5. Towards sustainability of the agro-food chains using LCA approaches

Over the last years, the interest in the environmental impacts associated with food systems has strongly

grown. Several works have confirmed the relative importance of "food and beverages consumption" in contributing to environmental impacts. For example, in a study carried out by Garnett (2008)it was calculated that food consumption in the UK contributes 19% of the UK's

greenhouse gas emissions. According to the United Nations Food and Agriculture Organisation (Steinfeld et al., 2006), livestock accounts for 18% of the global greenhouse gas emissions. Moreover, the Food and Agriculture Organisation (FAO) estimate that 3.3 Gt of CO_2 eq. is emitted owing to one third of food being wasted worldwide, making food wastage the third top GHG emitter after USA and China (FAO, 2013). In the EU, food consumption accounts for 20-30% of various environmental impacts and, in the case of eutrophication, more than 50% (Tukker et al., 2006).
Moreover, the global human population is deemed to grow by 34% in 2050; FAO predicts that a 70% increase of food production is going to be required in order to meet the needs of such population growth.

LCA provides a methodological framework, a supporting tool for evaluating and improving the environmental performance of food systems .The importance of such methodology in the agro-food sector is witnessed by several literature studies focusing on diverse agro-food products, some of which are listed in Table 4.

Table 4. Examples of LCA applications to the agro-food sector.

Food product	References
Vegetables	Jungbluth et al. (2000)
	Cellura et al. (2012a)
	Cellura et al. (2012b)
Wheat	Brentup et al. (2004)
	Charles et al. (2006)
	Meisterling et al. (2009)
Rice	Roy et al. (2005)
	Roy et al. (2007)
	Harada et al. (2007)
	Blengini and Busto (2009)
	Roy et al. (2009)
	Fusi et al. (2014)
Milk	Hospido et al. (2003)
	Cederberg & Mattsson (2000)
	Cederberg & Stadig (2003)
	Eide (2002)
	Haas et al. (2001)
	González-García et al. (2013b)
Meat	Jungbluth et al. (2000)
	Basset-Mens & van der Werf (2005)
	Cederberg & Stadig (2003)
Yogurt	González-García et al. (2013a)
Cheese	Berlin (2002)
Wine	Aranda et al. (2005)
	Ardente et al. (2006)
	Gonzalez et al. (2006)

	Rugani et al. (2009)
	Gazulla et al. (2010)
	Vázquez-Rowe et al. (2012)
	Benedetto (2013)
	Neto et al. (2013)
	Fusi et al. (2014)
Beer	Takamoto et al. (2004)
	Koroneos et al. (2005)
	Hospido et al. (2005)
	Cordella et al. (2008)
Oilve oil	Salomone and Ioppolo (2012)
Bread	Braschkat et al. (2003)
	Espinoza-Orias et al. (2011)
	Kulak et al. (2012)
Pasta	Notarnicola and Nicoletti (2001)
	Notarnicola et al. (2004)
	Salomone and Ciraolo (2004)
	Bevilacqua et al. (2007)
	Ruini et al. (2013)
Ready meals	Schmidt Rivera et al. (2014)

In the following section, a brief description of the LCA results found in literature regarding the main groups of food products, will be discussed.

LCA of vegetables

The environmental hotspots associated with the production of vegetables are of course influenced by the processes included in the LCA analysis: if a processing phase (washing, packaging) is considered within the boundaries of the study, such phase is going to have a consistent impact over the whole life cycle of the vegetable under investigation. When just the agricultural phase is under consideration, the
method of cultivation (greenhouse or open field, organic or conventional, and hydroponic or soil-based), variety and location of cultivation have a significant influence on the LCA results. As a general comment, it could be stated that, at the agricultural level, the filed operations for soil preparation as well as the use of fertilisers and pesticides are critical points of the environmental performance of vegetables. In addition, greenhouses, if used, do not play a secondary role in the environmental profile of agricultural products.

LCA of wheat and rice

The application of Life Cycle Assessment (LCA) to the agricultural production and cereals sector goes back to the beginning of the 21^{st} century, when some of the first LCA studies were performed. The main applications of LCA in the cereals sector have been devoted to different goals: identify the environmental hotspots in production systems performance, profile the environmental burden of production in a given area, compare the environmental burden of

different food products and different farming practices, as well as evaluate the environmental properties of a supply chain.
LCA studies performed on wheat can be distinguished into two main categories: studies addressing wheat as cereal, without indication of its final use or wheat used for bread production. Key issues in LCA on wheat are represented by the use of fertilisers, especially N fertilisers, and pesticides.
A further hot topic for cereals LCA - and wheat LCA in particular - is the comparison between different farming techniques, i.e. conventional vs organic farming or irrigated vs rain-fed farming. In the case of climate change it can be stated that, when conventional and organic wheat are transported the same distance to market, the organic wheat system produces less CO_2-eq per functional unit than the conventional wheat system. Of course, when other impacts categories are included in the assessment, same trade-offs can be expected between impact categories.

As previously discussed, rice cultivation is one of the main contributor of methane emissions in the atmosphere, making this activity relevant for global warming. Having said so, it is needless to say that one of the main hotspots of rice cultivation is represented by such emissions. Nevertheless, GHG emission is dependent on location, size of farms and the variety of rice. Other hotspots are represented by fuel consumption required for the mechanisation of field operations and the use of fertilisers, which strongly influence the acidification and eutrophication impact categories.

LCA of wheat derivates

Pasta and bread are the object of various LCA analyses. The majority of the studies adopted a cradle to grave approach, including in the analysis all life cycle phases, up to disposal. Besides the cultivation phase, which resulted to be determinant in all the studies carried out on pasta and bread, other stages of the life cycle, such as distribution and use, shown an "environmental importance". While for bread the impact of the consumption phase results to be significant depending on the consumer's behaviour (if bread is refrigerated or toasted), the use phase associated with pasta appears to be relevant in term of energy consumption and of the impacts connected to it. In some studies, also the production phase (pasta production and bread baking) and distribution are found to be critical.

LCA of wine and beer

Over the last years, the environmental impact associated with the wine production has been studied by several authors. These studies revealed the most critical stages of wine bottles over their life cycle: the grapes production (i.e. the agricultural phase) and the glass bottle production, while the distribution phase appears to be relevant only if long distances are covered (international distribution). The glass bottles production contributes to different environmental category such as global warming potential, abiotic depletion, acidification and eutrophication, being an highly energy-consuming process. The impact associated with the agricultural phase is mainly attributable to the use of fertilisers, pesticides as well as the diesel fuel required to carry out the field activities.

In the case of beer production, the stages of its life cycle having the highest impact appear to be: wort production followed by filtration and packaging and lastly fermentation and storage. In some cases, the bottle production, followed by packaging and beer production, was the subsystem that accounts for most of the emissions. The production and manufacturing of the packaging elements as well as the harvesting and transport of cereals were found to be responsible for the largest portion of such emissions.

LCA of dairy and meat products

The dairy industry has been studied extensively to determine its environmental impact in many European countries. Milk is one of the most important dairy products in European countries, and it has been reported that organic milk production can reduce pesticide use and mineral surplus in agriculture, but requires substantially more arable land than conventional production. The agricultural phase is reported to be the main hotspot in the life cycle of milk and semi-hard cheeses. Packaging, waste production and cleaning processes also have relevant impacts. The main environmental impacts associated with dairy processing are the high consumption of water, the discharge of effluent with high organic components and energy requirements.

Livestock production has a major impact on the environment. The impact associated with feed production, raising the livestock and manure handling are the greatest contributor to the overall impact resulting from meat production. The livestock sector increasingly competes for scarce resources, such as land, water, and energy, and has a severe impact on air, water and soil quality because of its emissions. For land use, energy use and climate change, the production of 1 kg of beef protein has the highest impact, followed by pork, whereas chicken has the lowest impact. This conclusion is based on results of the life cycle of meat production until the product left the farm gate. During the post-farm-gate stages of production of meat, such as processing, packaging, retail and household, there is an additional environmental impact; nevertheless small differences are expected in environmental impact of post-farm-gate stages among different meat products. Differences in environmental impact among pork, chicken, and beef can be explained mainly by three factors: utilization of nutrients and energy in feed, differences in enteric CH_4 emission between pigs and chicken, and cattle, and differences in reproduction rates. The environmental impacts of beef production system are reported to be dependent on the feed production and type of feed, animal housing and manure storage.

LCA of olive oil

According to the International Olive Oil Council (IOOC), olive oil is a typical Mediterranean product of great economic importance in the European Union, both in terms of production and consumption. It is mainly produced in the countries of the Mediterranean area, but new producing countries situated in America, Africa and Australia are gaining more and more relevance. The olive oil industry is responsible for diverse environmental impacts in terms of resource depletion, land degradation, air emissions and waste generation. These impacts may vary significantly as a result of the practices and techniques carried out in olive cultivation and oil production. Olive tree cultivation and the extraction of olive oil cause resource consumption, emissions into the air, water and soil, pruning and harvesting residues and huge quantities of waste that may have a great impact on land and water environments.

Generally, the most critical factors along the olive oil life cycle are (i) fertilisers and pesticides use along the agricultural phase, (ii) the glass bottle production required for the packaging phase. The milling phase appears to be a less relevant contributor to the overall environmental impact of olive oil.

LCA of ready meals

Economic growth, changing dietary habits and lifestyles will intensify environmental impacts of food in the future, mainly because of the increasing demand for convenience food in the developed world but also in China. The convenience food sector is expanding rapidly with the global ready-made meals market expected to grow by 3.2% by 2016. Currently, the US and the UK are the largest markets and the latter is expected to grow by 20% by 2017. A study undertaken in the UK, comparing ready ready-made meals and the same meals prepared at home, suggests that home-made meal are generally a better option than the ready-made meal for most environmental impacts. The main reasons for this are the avoidance of manufacturing, reduction in refrigerated storage and a lower amount of waste produced for the preparation of the home-made meal.

6. Scientific papers

Science of the Total Environment 472 (2014) 989–1000

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Delving into the environmental aspect of a Sardinian white wine: From *partial* to *total* life cycle assessment

Alessandra Fusi [a], Riccardo Guidetti [a], Graziella Benedetto [b,*]

[a] Dipartimento di Scienze Agrarie Ambientali, Produzione, Territorio, Agroenergia, University of Milano Italy, Via Celoria 2, 20133 Milano, Italy
[b] Department of Science for Nature and Environmental Resources, University of Sassari, Via Piandanna, 07100 Sassari, Italy

HIGHLIGHTS

- A 'cradle to grave' E-LCA was used for 750 ml bottle of a Sardinian white wine.
- The phases of vine planting, distribution and final disposal are included in the LCA.
- Hot-spots are glass bottle production and vine planting due to diesel consumption.
- The impact categories more affected by transport were AP, EP, POCP and GWP.
- Improvements were a lighter glass bottle or the use of polylaminate container.

ARTICLE INFO

Article history:
Received 8 September 2013
Received in revised form 29 November 2013
Accepted 30 November 2013
Available online 17 December 2013

Keywords:
Life cycle assessment
Winemaking
'Cradle to grave'
Sardinia

ABSTRACT

The aim of this study was to deepen the assessment of the environmental impacts of a white wine produced in Sardinia (FU 750 ml), performing an *attributional* LCA. The system boundaries were extended, from 'cradle to gate' (*partial* LCA) of a previous study, to 'cradle to grave' (*total* LCA), in order to identify the environmental impacts occurring along the wine life cycle stages (vine planting, grape production, wine production, bottling and packaging, distribution, final disposal of the glass bottle).
Some assumptions were made in order to quantify the environmental impact of the transportation phase, regarding the few data which were available.
Inventory data were mainly collected through direct communication with the Company involved in the study. Results showed that the environmental performance of wine was mostly determined by the glass bottle production (for all impact categories except ozone layer depletion). The second contributor was the agricultural phase, which included two sub-phases: vine planting and grape production. Results showed that the vine planting subphase was not negligible given its contribution to the agricultural phase, mainly due to diesel fuel consumption. Transportation impact was found to be relevant for long distance distribution (USA); the impact categories more affected by transport were acidification, eutrophication, photochemical oxidation and global warming potential. Suggested opportunities to reduce the overall environmental impact were the introduction of a lighter glass bottle or the substitution of the glass bottle with a polylaminate container.

1. Introduction

Over the past two decades, the worldwide awareness regarding environmental issues has consistently increased: consumers are now changing their behaviour to integrate environmental considerations into lifestyle choices. The environmental aspect is now one of the variables taken into consideration by consumers during the purchasing process. In some cases, consumers are willing to pay a premium for environmentally friendly products (Barber et al., 2009). As mentioned by Berners-Lee et al.'s (2011) businesses of all sizes are increasingly looking to modify their actions to manage their impact, to protect their reputations and to prepare for tighter regulations. Over the last few years, the evaluation and communication of products with environmental impacts, by means of an *eco-label*, are starting to gain ground within the agro-food sector. The farm-gate approach has the advantage of encouraging the use of best practices in each production stage, allowing, on the one hand, the reduction of emissions which are directly controlled by the farmer and, on the other hand, the creation of policies that are applicable at the company level (Dick et al., 2008). The agricultural sector is considered, after fossil fuels, the main cause of greenhouse gas emissions. According to the last published IPCC report, the agricultural sector is the second responsible for global GHG production, emitting between 5.1 and 6.2 Gt CO_2 eq., which corresponds to the 10–

* Corresponding author. Tel.: +39 079 229355, +39 331 7351805 (mobile); fax: +39 079 229356.
E-mail addresses: alessandra.fusi@unimi.it (A. Fusi), riccardo.guidetti@unimi.it (R. Guidetti), gbenedet@uniss.it (G. Benedetto).

0048-9697/$ – see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.scitotenv.2013.11.148

12% of total anthropogenic GHG emissions and, including forestry activities, and is responsible for 50% and 70% of methane (CH_4) and nitrous oxide (N_2O) emissions, respectively, and for 25% of the carbon dioxide production (CO_2) (Smith et al., 2007).

This is due to direct emissions deriving from agricultural operations, for example carbon dioxide (CO_2) emissions from the use of diesel by tractors and irrigation equipment or emissions from agricultural inputs used (e.g. fertiliser, herbicides and pesticides).

However, it is important to consider indirect emissions generated off-farm as a result of the manufacturing of inputs used on the farm, for example GHG emissions from the use of natural gas in the production of commercial fertiliser and chemicals. As stated by Coderoni and Bonati (2013), the agricultural sector will represent one third of European emissions by 2050 (if *decarbonisation* of other sectors has not yet occurred). Therefore, the relevance of the agricultural sector in climatic policy is expected to increase.

The wine sector must deal with this scenario, and emissions associated with both the productive phase and the distribution phase must be considered. With regard to the latter, it must be highlighted that wine largely contributes to the global agro-food trade: in 2011, 10 million t of wine were exported all over the world and wine is the 19th highest agro-food product exported by quantity and the 7th by value (1000$) (FAOSTAT).

The whole supply chain must therefore be considered in order to take into account all impacts deriving from wine production, as suggested for other agro-food sectors (Iribarren et al., 2010; Berners-Lee et al., 2011).

Protocols for the evaluation of wine emissions are currently being set up by important Institutions, like the International Organisation of Vine and Wine (OIV).[1] Therefore, the whole industry must address this issue in the very near future, private companies included.

Italy is one of the leading wine producing countries, with more than 42 million hl produced in 2011 (OIV (International Organisation of Vine and Wine), 2013), and plays a dominant role among the traditional exporting countries, according to FAO data (2011), accounting for 23% of global wine exports. The UK, the United States and Germany are its main buyers (Istat-Coeweb, 2010).

Nowadays, we are witnessing the rise of a “green competition” in the international wine trade. Environmental issues are in fact not only popular both in traditional wine importing countries (e.g. the United Kingdom) and in global larger markets (e.g. the United States), but also in countries that have been recently gaining a share of the export market (e.g. Australia, New Zealand and South Africa) and in traditionally net exporting countries (e.g. France).

Hence, there is interest in analysis that assesses the environmental impacts linked to the production of an Italian bottle of wine that is widely sold in markets where the interest on environmental issues is growing and other competitors are embracing environmental management systems (as Hardie (2000) pointed out, Australian wine producers).

In this context, it is important to be able to assess the environmental load linked to wine production. As mentioned by Zamagni et al. (2012), the European Commission states that “LCAs provide the best framework for assessing the potential environmental impacts of products currently available” (CEC, 2003). LCAs might be conducted by an industry sector in order to identify areas where improvements can be made, in environmental terms. In recent years, a number of major companies have cited LCAs in their marketing and advertising, to support claims that their products are ‘environmentally friendly’[2] or even ‘environmentally superior’ to those of their rivals (World Resource Foundation).

[1] Pattara et al. (2012b) used the OIV guidelines as a methodological basis in their cradle to gate study.

[2] Examples are Soave Consortium in Italy and Taylors Wines in South Australia (Lambert, 2010).

Life cycle assessment (LCA) is a standardised methodology used for estimating the environmental burdens associated with the life cycle of products or processes (ISO, 2006a,b). This methodology is considered to be effective for evaluating environmental performance in the food and beverage sector (Andersson, 1998; Cerutti et al., 2011; González-García et al., 2013a,b), and, of course, in the viticulture and vinification sectors. Several studies have been carried out in order to assess the environmental performance of wine using the life cycle assessment approach (e.g. Zabalza et al., 2003; Notarnicola et al., 2003; Aranda et al., 2005; Montedonico, 2005; Gonzalez et al., 2006; Ardente et al., 2006; Petti et al., 2006; Rugani et al., 2009; Carta, 2009; Colman and Päster, 2009; Schlich, 2010; Petti et al., 2010; Gazulla et al., 2010; Barry, 2011; Bosco et al., 2011; Pattara et al., 2012a; Point et al., 2012; Vázquez-Rowe et al., 2012; Comandaru et al., 2012; Neto et al., 2013; Vázquez-Rowe et al., 2013; Benedetto, 2013; Villanueva-Rey et al., 2013).

As can be seen from Rugani et al. (2013), some of them adopted a ‘cradle to grave’ perspective, with the inclusion of the distribution phase (e.g. Gazulla et al., 2009; Point et al., 2012; Neto et al., 2013), while others preferred a ‘cradle to gate’ approach, without taking into consideration the distribution (e.g. Vázquez-Rowe et al., 2012; Benedetto, 2013). With some exceptions (Montedonico, 2005; Pizzigallo et al., 2008; Rugani et al., 2009; Carta, 2009; Bosco et al., 2011; Comandaru et al., 2012; Benedetto, 2013), the vine planting phase represents a stage of the wine life cycle that is rarely considered in wine LCA studies due to a lack of data.

According to these studies, the production of glass bottles and the viticulture phase are environmentally relevant in the overall wine life cycle.

Over the years, more and more importance has been given to the assessment of the life cycle as a whole: therefore, the interest has shifted from *partial*[3] to *total*[4] LCA, as already outlined on another occasion (Benedetto et al., 2013). For this reason, this study proposes the evaluation of environmental impacts associated with the production of a white wine produced in Sardinia by Sella & Mosca, including additional stages of the production process compared to a previous study (Benedetto, 2013).

The aim of this study was to deepen the assessment of the environmental impacts of a white wine produced in Sardinia (FU 750 ml), performing an *attributional* LCA. The system boundaries were extended from ‘cradle to gate’ (*partial* LCA) to ‘cradle to grave’ (*total* LCA), in order to identify the environmental impacts occurring along the wine life cycle stages (vine planting, grape production, wine production, bottling and packaging, distribution, final disposal of the glass bottle). The analysis was performed on Vermentino wine produced by one of the biggest companies in Europe (Sella & Mosca), which exports its wine all over the world, won the award for the Eco-friendly winery in 2012 and was named winery of the year in 2013 in the Gambero Rosso Guide. This company, founded more than one century ago, has more than 550 ha of vineyard and produces approximately 7 million bottles per year; the production of Vermentino “La Cala”, which was selected for this study because it represents a flagship product of the company's portfolio, amounts to 500,000 bottles per year.

[3] The PEF Guide (2012) specifies that ‘cradle to gate’ is “a partial product supply chain, from the extraction of raw materials (cradle) up to the manufacturer's “gate”. The distribution, storage, use stage and end-of-life stages of the supply chain are omitted” (p. 75); the ‘gate to gate’ and ‘gate to grave’ LCAs are also partial (p. 76). The same definition is included in the ENVIFOOD Protocol Environmental Assessment of Food and Drink Protocol (2012, p. 13).

[4] ‘Cradle to grave’ LCA is referred to: “a product's life cycle that includes raw material extraction, processing, distribution, storage, use, and disposal or recycling stages. All relevant inputs and outputs are considered for all of the stages of the life cycle” (PEF Guide, p. 75); in the ENVIFOOD Protocol, this definition is reported for the cradle to grave inventory as “a complete life cycle of a product which includes all the consecutive and interlinked stages of a product system from material acquisition through to end-of-life” (p. 13).

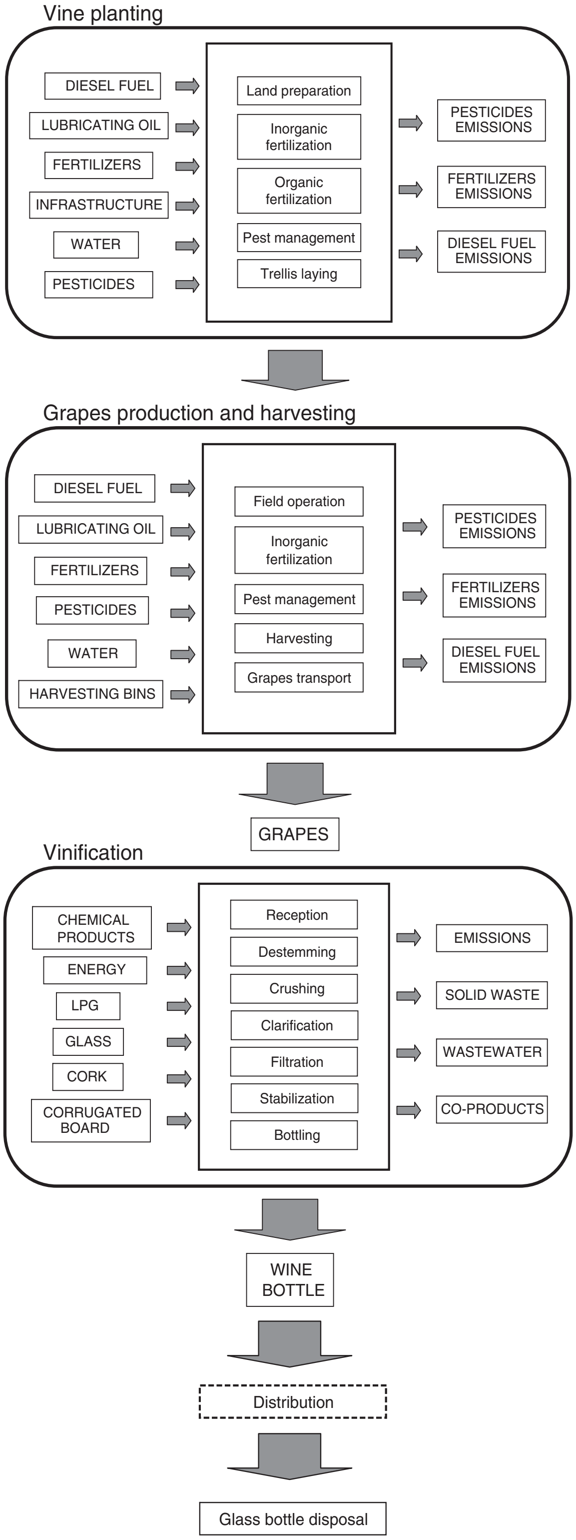


Fig. 1. System boundaries.

Table 1
Data inventory for vine planting (data related to FU).

	Units	
Inputs		
Diesel fuel	g	69.51
Lubricating oil	g	2.08
Compost	g	2.64
Potassium chloride	g	1.21
Fertiliser P_2O_5	mg	396.72
Steel	g	3.91
Concrete	g	74.45
Fungicides[a]	mg	33.46
Pesticides[b]	mg	314.5
Dinitroaniline-compounds[c]	mg	2.08
Cyclic N-compounds[d]	mg	2.98
[Sulfonyl]urea-compounds[e]	mg	7.93
Acetamide-anillide-compounds[f]	mg	6.45
Glyphosate	mg	21.82
Dithiocarbamate-compounds[g]	mg	65.03
Water	m^3	0.01
Outputs		
Vineyard	m^2	1
Emissions to air		
Carbon dioxide (diesel)	g	11.71
Carbon monoxide (diesel)	mg	64.79
Particulate	mg	9.74
Hydrocarbons	mg	12.77
Nitrogen oxides	g	0.11
Dinitrogen monoxide (fertiliser)	mg	2.07
Ammonia (fertiliser)	mg	22.48
Glyphosate	mg	2.18
Dinitrophenol	mg	0.49
Mancozeb	mg	3.24
Morpholine	mg	0.30
Dimethomorph	mg	0.57
Benzophenone	mg	0.20
Metiram	mg	3.26
Pyraclostrobin	mg	0.21
Carfentrazone-ethyl	mg	0.30
Sulphur	g	0.025
Metalaxyl-M	mg	0.65
Copper oxychloride	mg	4.68
Quinoline	mg	0.83
Copper oxide	mg	0.89
Emissions to water		
Nitrate (fertiliser)	g	0.12
Phosphate (fertiliser)	mg	7.03
Glyphosate	mg	2.18
Dinitrophenol	mg	0.49
Mancozeb	mg	3.24
Morpholine	mg	0.30
Dimethomorph	mg	0.57
Benzophenone	mg	0.20
Metiram	mg	3.26
Pyraclostrobin	mg	0.21
Carfentrazone-ethyl	mg	0.30
Sulphur	g	0.025
Metalaxyl-M	mg	0.65
Copper oxychloride	mg	4.68
Quinoline	mg	0.83
Copper oxide	mg	0.89
Emissions to soil		
Glyphosate	mg	16.35
Dinitrophenol	mg	3.71
Mancozeb	mg	24.32
Morpholine	mg	2.23
Dimethomorph	mg	4.26
Benzophenone	mg	1.48
Metiram	mg	24.45
Pyraclostrobin	mg	1.56
Carfentrazone-ethyl	mg	2.23
Sulphur	g	0.19
Metalaxyl-M	mg	4.84
Copper oxychloride	mg	35.06
Quinoline	mg	6.21
Copper oxide	mg	6.69

2. Material and methods

2.1. Goal and scope definition

The environmental performances of a Sardinian wine (Vermentino) were assessed using the LCA methodology.

The selected functional unit (FU) was a bottle (750 ml) of Vermentino white wine produced by a large company, Sella & Mosca.

The studied system (Fig. 1) was not restricted to the wine making process but also included the agricultural phases of vine planting and grape production, as well the final disposal of packaging (glass bottle).

The present study was carried out at two levels:

1 Level A ignored the wine transportation phase;

2 Level B included the wine transportation phase.

2.2. Inventory

Data concerning field operations and the wine making process were directly obtained from the company involved in the analysis. Foreground data were integrated with database information (Ecoinvent version 2.2 (Frischknecht et al., 2007); LCA Food DK (Nielsen et al., 2003)).

Data concerning grape production and wine making processes refer to 2012, which can be considered an average year since grape production has remained approximately constant over the last 5 years.

2.2.1. LCI of agricultural phase

The vine planting phase lasts for approximately three years, after which the vineyard starts being productive. A vineyard is considered to be productive for 27 years (information provided by the company), and vines at the end of their lifetime are used to produce energy. The analysis carried out did not take into account the end of life of the vineyard.

Table 1 reports a data inventory for the vine planting phase; Table 2 shows a data inventory for the grape production phase.

For the agricultural phase (both for vine planting and grape production), the emissions due to fertiliser were included: nitrogen emissions to air and water (dinitrogen monoxide (direct and indirect emissions), ammonia and nitrate) were computed using the IPCC (2006a,b) (a) emission factors; phosphate emissions were calculated in accordance with Smil (2000). Since grape stalk is spread onto the field as a fertiliser, the emissions due to its use were also computed. The nitrogen content in grape stalk was calculated in accordance with Rossini et al. (2010).

In order to calculate pesticide emissions precisely, it is necessary to have data regarding, among others, the way in which a pesticide is applied and the meteorological conditions during application (EMEP/EEA, 2013). Since all of these data were not available, the emission factors used can be considered as first estimates. Pesticide emissions into the air, water and soil were estimated in accordance with Margni et al. (2002) and Audsley (1997). According to these studies, the fraction of active ingredient entering the soil is assumed to be 85% of the total applied quantity; 5% remains on the plant and 10% is emitted into the air. The run-off of the active ingredient from the soil into the water is assumed to be a maximum of 10% of the applied dose. Regarding fuel use, the emissions that each machine generates for field operations

Notes to Table 1:

[a] Meptyldinocap, spiroxamina, dimethomorph, metrafenone, quinoxifen, spirotetramat 48.

[b] Sulphur, copper oxychloride, copper hydroxide.

[c] Pyraclostrobin.

[d] Carfentazone-etile.

[e] Flufenoxuron.

[f] Metalaxil M.

[g] Mancozeb, metiram.

Table 2
Data inventory for grape production (data related to FU).

	Units	
Inputs		
Diesel fuel	g	12.1
Lubricating oil	g	0.36
Fertiliser (N)	mg	13.5
Fertiliser (P_2O_5)	mg	83
Fertiliser (K_2O)	g	0.11
Fungicides[a]	g	0.34
Pesticides[b]	g	3.17
Dinitroaniline-compounds[c]	mg	21
Cyclic N-compounds[d]	mg	30
[Sulfonyl]urea-compounds[e]	mg	80
Acetamide-anillide-compounds[f]		65
Glyphosate	g	0.22
Dithiocarbamate-compounds[g]	g	0.66
HDPE (bins)	mg	46.28
Water	m^3	0.10
Land	m^2	1
Outputs		
Product		
Grapes	kg	1.071
Emissions to air		
Carbon dioxide (diesel)	g	38
Carbon monoxide (diesel)	g	0.26
Particulate	mg	40
Hydrocarbons	mg	50
Nitrogen oxides	g	0.38
Dinitrogen monoxide (fertiliser)[h]	mg	8.09
Ammonia (fertiliser)[h]	mg	90
Glyphosate	mg	23
Dinitrophenol	mg	5
Mancozeb	mg	32.5
Morpholine	mg	3
Dimethomorph	mg	5.75
Benzophenone	mg	2
Metiram	mg	33
Pyraclostrobin	mg	2
Carfentrazone-ethyl	mg	3
Sulphur	g	0.26
Metalaxyl-M	mg	6.5
Copper oxychloride	mg	30
Quinoline	mg	4
Copper oxide	mg	9
Emissions to water		
Nitrate (fertiliser)[h]	g	0.48
Phosphate (fertiliser)	mg	1
Glyphosate	mg	23
Dinitrophenol	mg	5
Mancozeb	mg	32.5
Morpholine	mg	3
Dimethomorph	mg	5.75
Benzophenone	mg	2
Metiram	mg	33
Pyraclostrobin	mg	2
Carfentrazone-ethyl	mg	3
Sulphur	g	0.26
Metalaxyl-M	mg	6.5
Copper oxychloride	mg	30
Quinoline	mg	4
Copper oxide	mg	9
Emissions to soil		
Glyphosate	g	0.17
Dinitrophenol	mg	37.5
Mancozeb	g	0.24
Morpholine	mg	22.5
Dimethomorph	g	43.13
Benzophenone	mg	15
Metiram	g	0.24
Pyraclostrobin	mg	15
Carfentrazone-ethyl	mg	22.5
Sulphur	g	1.95
Metalaxyl-M	mg	48.75
Copper oxychloride	g	0.23
Quinoline	mg	30
Copper oxide	mg	67.5

Table 3
Data inventory for wine making (data related to FU).

	Units	
Inputs		
Harvested grapes	kg	1.071
Liquid sulphur dioxide	mg	64.3
Liquid nitrogen	g	0.37
Bentonite	g	0.35
LPG	g	0.64
Water (tap)	kg	5.34
Electricity	MJ	0.4
Outputs		
Products		
Wine	l	0.75
Marc and lees	kg	0.27
Stalks	kg	0.05
Emissions		
Ethanol (fermentation)	g	0.165
Carbon dioxide (LPG)	g	1.87
Dinitrogen monoxide (LPG)	mg	0.003
Methane (LPG)	mg	0.03
Waste		
Wastewater	kg	5.34

were estimated using data from the Swiss Federal Office for the Environment (DETEC (Federal Department of the Environment, Transport, Energy and Communications)).

No change in the overall soil carbon content was assumed because the fields were previously dedicated to vine cultivation.

2.2.2. LCI of wine making phase

Carbon sequestration by grape vines and the subsequent release of CO_2 during fermentation were excluded from the analysis (Notarnicola et al., 2003; Carta, 2009; Benedetto, 2013; Rugani et al., 2013). Moreover the PEF Guide reports that "credits associated with temporary (carbon) storage or delayed emissions shall not be considered in the calculation of the default EF impact categories" (p. 36). On the other hand, emissions of ethanol were included as they are known to contribute to photochemical oxidation (Notarnicola et al., 2003; Pizzigallo et al., 2008; Vázquez-Rowe et al., 2012; Point et al., 2012; Neto et al., 2013). Ethanol emissions were estimated using the United States Environmental Protection Agency (USEPA) (1995) emission factor.

Air emissions associated with liquefied petroleum gas (LPG) use were estimated using emission factors from IPCC (2006b).

Table 3 reports a data inventory for the wine making phase.

2.2.3. LCI for the bottling and packaging phase

Since this phase takes place within the winery, it was not possible to acquire specific data regarding electricity use for the bottling and packaging subsystem. Therefore, the entire electricity consumption was assigned to the wine making stage based on the assumption that only a small proportion of the total energy is attributable to bottling and packaging (Guidetti, 2005; Bosco et al., 2011).

The same considerations were made for water use. Table 4 reports the inventory of the bottling and packaging phase.

Notes to Table 2:
a Meptyldinocap, spiroxamina, dimethomorph, metrafenone, quinoxifen, spirotetramat 48.
b Sulphur, copper oxychloride, copper hydroxide.
c Pyraclostrobin.
d Carfentazone-etile.
e Flufenoxuron.
f Metalaxil M.
g Mancozeb, metiram.
h Emissions due to the spread of grape stalk were included.

Table 4
Data inventory for bottling and packaging (data related to FU).

	Units	
Inputs		
Wine	l	0.75
White glass	kg	0.56
Cork	g	3.5
Paper for labels	g	1
Corrugated board	g	66.7
Outputs		
Product		
Bottle of wine	p	1
Waste		
Glass	g	14

White glass bottles were assumed to be manufactured out of approximately 61% recycled glass. This value was retrieved from the Ecoinvent database (version 2.2) for white packaging glass.

2.2.4. Glass bottle disposal

Different waste scenarios for glass bottle were considered, coherently with distribution destinations; therefore, three waste scenarios were chosen: Italian, European and American. In Italy, 34% of the glass is landfilled and 66% is recycled (Co.Re.Ve.), while in Europe an average of 32% of the glass is landfilled and 68% is recycled (FEVE (The European Container Glass Federation)), and 72% of the glass is landfilled and 28% is recycled in the United States (CRI (Container Department Institute)).

2.2.5. Allocation

During the wine making process, other products besides wine are produced: marc, lees and stalks. Marc and lees are sold to a distillery; stalks are, as previously mentioned, spread on the field.

An allocation was made on an economic basis since the economic value best reflects the relative importance of the different co-products within the wine industry (Gazulla et al., 2010). Table 5 reports the economic allocation factors used, as well as the mass share of each co-product, as indicated by the company.

2.2.6. Transport

To date, few studies have included the distribution phase in wine life cycle assessments; among them, the following could be cited: Aranda et al. (2005); Ardente et al. (2006); Gonzalez et al. (2006); Petti et al. (2006); CIV (2008); Gazulla et al. (2010); Barry (2011); Bosco et al. (2011); Point et al. (2012); and Burja and Burja (2012). Since transport can be relevant in the overall environmental impact of wine (Colman and Päster, 2009; Saxe, 2010; OIV (International Organisation of Vine and Wine), 2013), it was decided to include this phase in our study, following the guidelines established by the Product Environmental Footprint (PEF) Guide (Manfredi et al., 2012).

Wine bottles are distributed within national borders and abroad. Due to the lack of information regarding international distribution, the transportation phase was neglected in the Level A analysis, with the aim of not reducing the reliability of the results.

Table 5
Allocation factors and mass share.

Product	Mass %	Economic allocation factor (%)
Wine	70	99.95
Marc and lees	25	0.05
Stalk	5	0

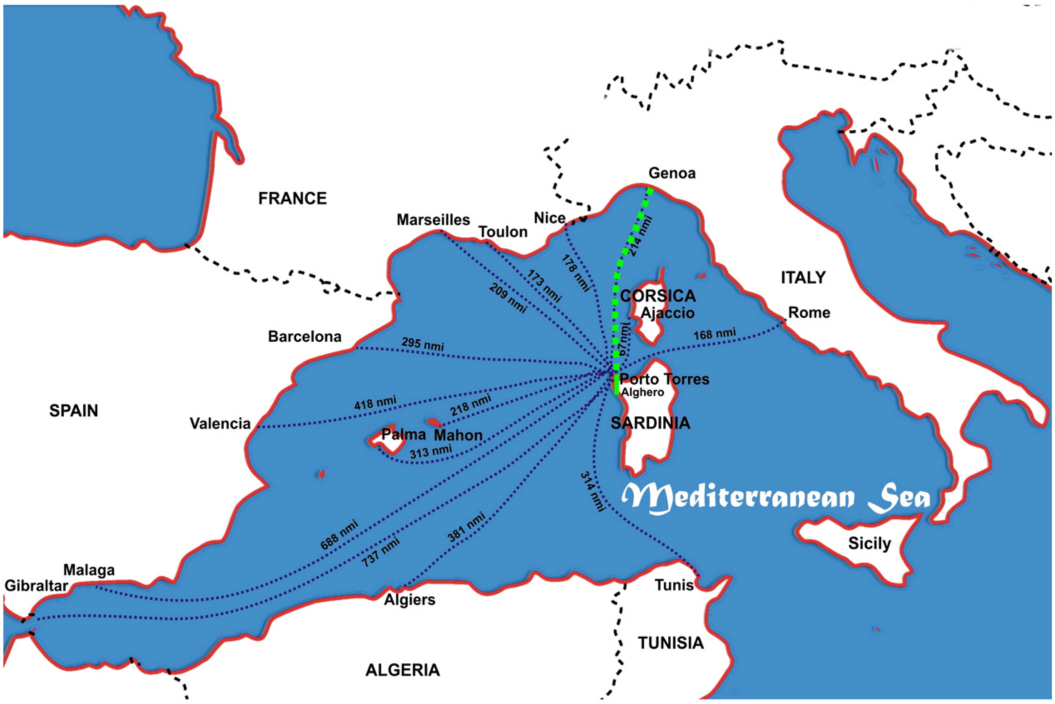


Fig. 2. Itinerary from Alghero to Porto Torres (by road) and from Porto Torres to Genoa (by sea). Common itinerary for all distribution destinations outside Sardinia. Source: Autorità Portuale di Olbia e Golfo Aranci

Some assumptions were made in order to perform the Level B analysis, in order to have an estimation of the transportation phase impacts, and their role in the overall wine life cycle assessment.

Apart from Sardinian distribution, which was performed only by road transportation, the distribution in all other destinations (within national borders (the Italian peninsula), and abroad) presented two common aspects (Fig. 2):

1 Transportation by road from Alghero (where the company is located) to Porto Torres;
2 Transportation by sea from Porto Torres to Genoa.

As for national distribution (Sardinia and the Italian peninsula), the specific destination (final destination or distribution centre) was available. Thus, it was possible to calculate the accurate amount of kilometres covered by the product both overland and by sea.

Information regarding international distribution was limited to destination countries and transportation means (Autorità Portuale di Genova). Therefore, it was possible to calculate the accurate number of kilometres covered by the product overland and by sea only until the Genoa port. Since no other precise information (from Genoa to the final destinations) was available, it was decided to calculate an "average point" of destination within each country. Instead of taking as a destination point the capital of the country or a place in the middle of the country, it was decided to estimate an average destination place, calculated by taking into account the distance from Genoa to the ten most populated cities of each country (Office for National Statistics; FSO (Federal Statistical Office); Statistical Offices of the Länder). Therefore, the

Table 6
Distribution destinations and transportation means.

				Transportation means	% total production
National distribution	Sardinia	Cagliari	Lorry > 32 t	9.7	30
		Carbonia-Iglesias		0.3	
		Nuoro		0.2	
		Ogliastra		0.1	
		Oristano		1.6	
		Olbia-Tempio		4.9	
		Sassari		5.9	
		Medio Campidano		7.2	
	Peninsula	Frascati (Roma)[a]	Lorry > 32 t, container ship	6.8	22
		Canale (Cuneo)[a]		15.2	
National distribution	Europe	England	Lorry > 32 t, container ship	5.5	48
		Germany		38.3	
		Switzerland		4	
	United States		Lorry > 32 t, container ship	52.2	

[a] Distribution centre.

Table 7
Parameters and respective changes considered in the sensitivity analysis.

		Default value	Range Min	Range Max
		kg N_2O-N/kg N		
Fertiliser used	N_2O emission factor from all N inputs (direct emissions)	0.01	0.003	0.03
	N_2O emission factor from N volatilization and re-deposition	0.01	0.002	0.05
	N_2O emission factor from leaching	0.0075	0.0005	0.025
		Share of N which is transferred		
	Volatilization for synthetic fertiliser	0.10	0.03	0.30
	Volatilization for organic fertiliser	0.20	0.05	0.50
	N losses by leaching	0.30	0.10	0.80
LPG used		mg/MJ		
	CO_2 emission factor	63,100	61,600	65,600
	CH_4 emission factor	1	0.3	3
	N_2O emission factor	0.1	0.03	0.3

destination point obtained represents an average place within the most populated cities of each country considered (Supplementary data: Tables 1, 2 and 3). The assumption made as the basis of this decision was as follows: imported wine is more likely to be in demand in cities with a higher number of inhabitants.

As for U.S. distribution, nautical miles from Genoa to New York port were calculated; New York port was selected, among all American ports, as the most connected to the port of Genoa (Genoa Port Authority Official Data, 2012). Due to the lack of information, the U.S. distribution was only considered until the port of New York; transport from the port to other possible destinations was excluded. Table 6 lists the destinations of wine bottles, both within national borders and abroad, and the transportation means considered in the analysis in accordance with the information provided by the company.

2.3. Impact assessment

SimaPro (version 7.3.2) was used to model the life cycle of Vermentino wine. Consistently with other studies (Aranda et al., 2005; Petti et al., 2010; Gazulla and Raugei, 2010; Vázquez-Rowe et al., 2012; Point et al., 2012; Benedetto, 2013), the following impact categories were selected to evaluate the environmental impact of the wine under study: global warming potential (GWP), acidification potential, eutrophication potential, photochemical ozone creation potential, ozone layer depletion (ODP) and abiotic depletion. LCIA was carried out using the CML baseline 2000 method (Guinée et al., 2002).

2.4. Sensitivity analysis

A set of parameters was changed and its influence on the results was evaluated. The most uncertain parameters were taken into account to run the sensitivity analysis. Consistently with Neto et al. (2013), in the agricultural phases, the parameters associated with the emission of nitrogen compounds due to fertiliser use were considered. For the wine production phase, the emission factors of carbon dioxide, methane and

Table 8
Results (expressed in absolute values and in percentage of contribution) from the characterisation step presented for each impact category.

Impact categories	Units	Agricultural phase Value	Agricultural phase % over total	Wine making phase Value	Wine making phase % over total	Bottling and packaging Value	Bottling and packaging % over total	Total value
Abiotic depletion	kg Sb eq.	2.57E−03	34.19	7.61E−04	10.10	4.19E−03	55.71	7.53E−03
Acidification	kg SO_2 eq.	1.52E−03	22.13	8.46E−04	12.29	4.51E−03	65.58	6.88E−03
Eutrophication	kg PO_4-eq.	3.22E−04	35.75	1.67E−04	18.55	4.12E−04	45.70	9.02E−04
Global warming (GWP100)	kg CO_2 eq.	1.69E−01	16.86	2.74E−01	27.23	5.62E−01	55.91	1.01E+00
Ozone layer depletion (ODP)	kg CFC-11 eq.	1.58E−07	71.04	5.51E−09	2.48	5.89E−08	26.48	2.23E−07
Photochemical oxidation	kg C_2H_4 eq.	7.54E−05	28.12	9.74E−05	28.31	1.60E−04	49.77	3.44E−04

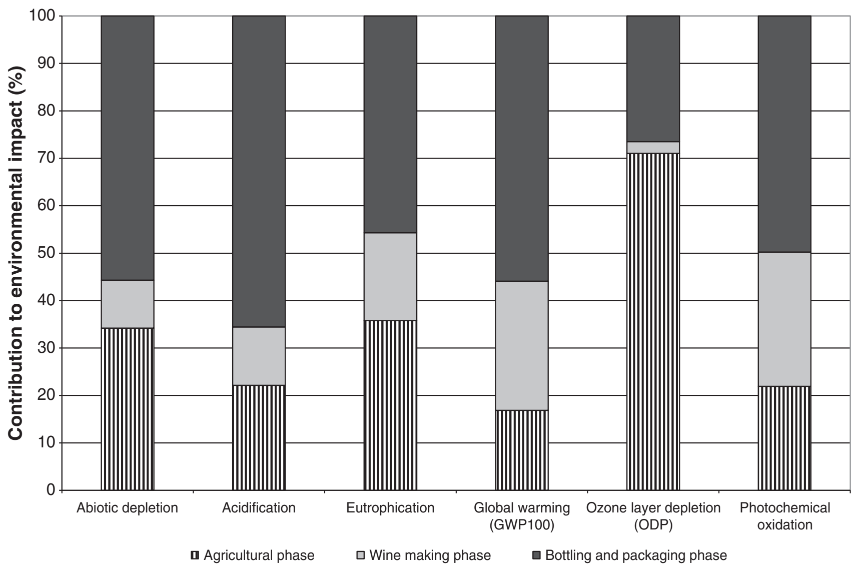


Fig. 3. Contribution of each phase (agricultural, wine making and bottling and packaging phase) to produce one bottle of wine.

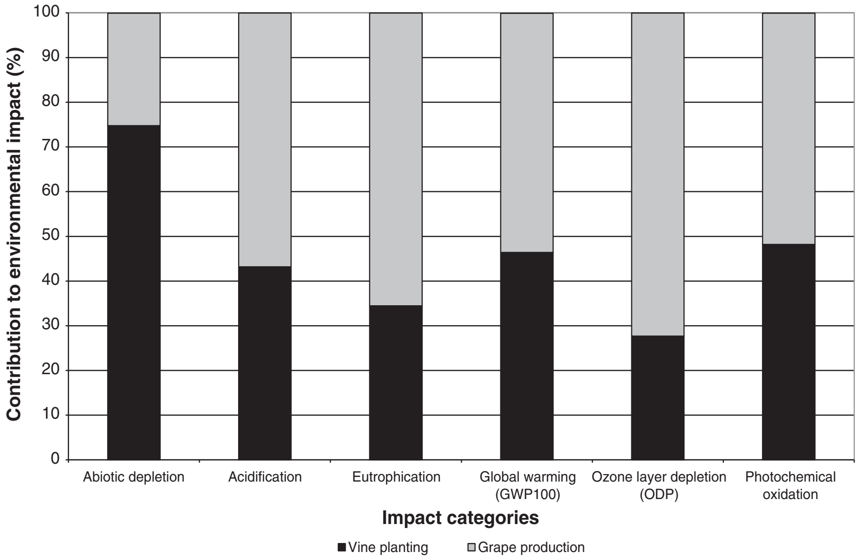


Fig. 4. Contribution of vine planting and grape production to the agricultural phase impact.

dinitrogen monoxide related to the use of LPG were changed. The emission factors both for fertiliser emissions and LPG emissions were modified within the range defined in the IPCC (2006a,b) (minimum and maximum value). The analysis was carried out considering, firstly, all minimum values of emission factors and, secondly, all maximum values (Table 7).

3. Results and discussion

3.1. Level A

Table 8 reports the total and relative impact values per FU linked to the three processes under study: the agricultural phase (vine planting and grape production), the wine making phase and the bottling and packaging phase. Fig. 3 shows the contribution of each phase to the production of one bottle of wine. The wine bottle disposal accounted for less than 0.5% of the overall environmental impact for all of the impact categories, so it was not considered a relevant process.

The bottling and packaging phase represented the main contributor to all impact categories except for ozone layer depletion, for which the agricultural phase played the most important role. The principal carrier of environmental impact was glass bottle production, for all impact categories (consistent with the results obtained by Petti et al. (2006)).

Although the system boundaries were changed compared to the previous study (Benedetto, 2013), the use of glass bottles remains a key node in the company production process.

As for the agricultural phase and the wine making phase, the following considerations could be made. Abiotic depletion is due to the consumption of fossil-based energy resources, mainly used in the agricultural phase (as diesel fuel) and, secondly, in the wine making phase (as electricity consumption and LPG use). Acidification, which is mostly related to the emission of SO_2 and NO_x to air, was, for the major part, caused by the use of electricity and diesel fuel and by diesel combustion for agricultural operations. Eutrophication was primarily associated with emissions due to fertiliser use in the agricultural phase and with wastewater produced during the wine making process. With regard to GWP, the main contributors were diesel fuel production and consumption (agricultural phase) and electricity consumption (wine making phase). ODP impacts were primarily associated with the emissions related to the production of pesticides used in the agricultural phase. For photochemical oxidation, the contributions of the agricultural phase (due to diesel fuel and pesticide production) and the wine making phase (due to ethanol emissions during the fermentation process) were similar.

With respect to other studies (Neto et al., 2013; Vázquez-Rowe et al., 2012), the contribution to the overall impact assessment of the agricultural phase and the wine making phase was lower for all impact categories except ODP (which was found to be consistent). On the other hand, the burden of bottling and packaging was higher.

The lower amount of fertilisers used and wastewater produced in the present study may have determined a reduction of the eutrophication associated with agricultural and wine making phases. The same consideration could be stated for abiotic depletion, acidification and GWP; in these cases, the inputs involved were electricity and LPG (lower with respect to the above mentioned studies) and diesel fuel consumption (lower with respect to the study carried out by Neto et al. (2013)). As for the GWP value (1.01 kg CO_2-eq./bottle), it was found to be consistent with the results obtained by Ardente et al. (2006), Gazulla et al. (2010) and Bosco et al. (2011), which lie between 0.6 and 1.3 kg CO_2-eq./bottle, and the results obtained in other wine-related studies (Vázquez-Rowe et al., 2013) in which the GHG emissions per bottle were between 0.65 and 1.17 kg CO_2-eq.

As already specified, the agricultural phase of the present study included two sub-phases: vine planting and grape production. The

Table 9
Sensitivity analysis results, calculated for the characterisation step, expressing the changes for each impact category with respect to the reference case.

Impact categories	Agricultural phase	Wine making phase	Bottling and packaging	Total variation %
	Variation %	Variation %	Variation %	
Abiotic depletion	0	0	0	0
Acidification	−6.51, +19.13	0	0	−1.44, +4.20
Eutrophication	−19.13, +50.78	0	0	−6.84, +18.20
Global warming (GWP100)	−1.79, +7.08	−0.02, +0.03	0	−0.31, +1.20
Ozone layer depletion (ODP)	0	0	0	0
Photochemical oxidation	0	0	0	0

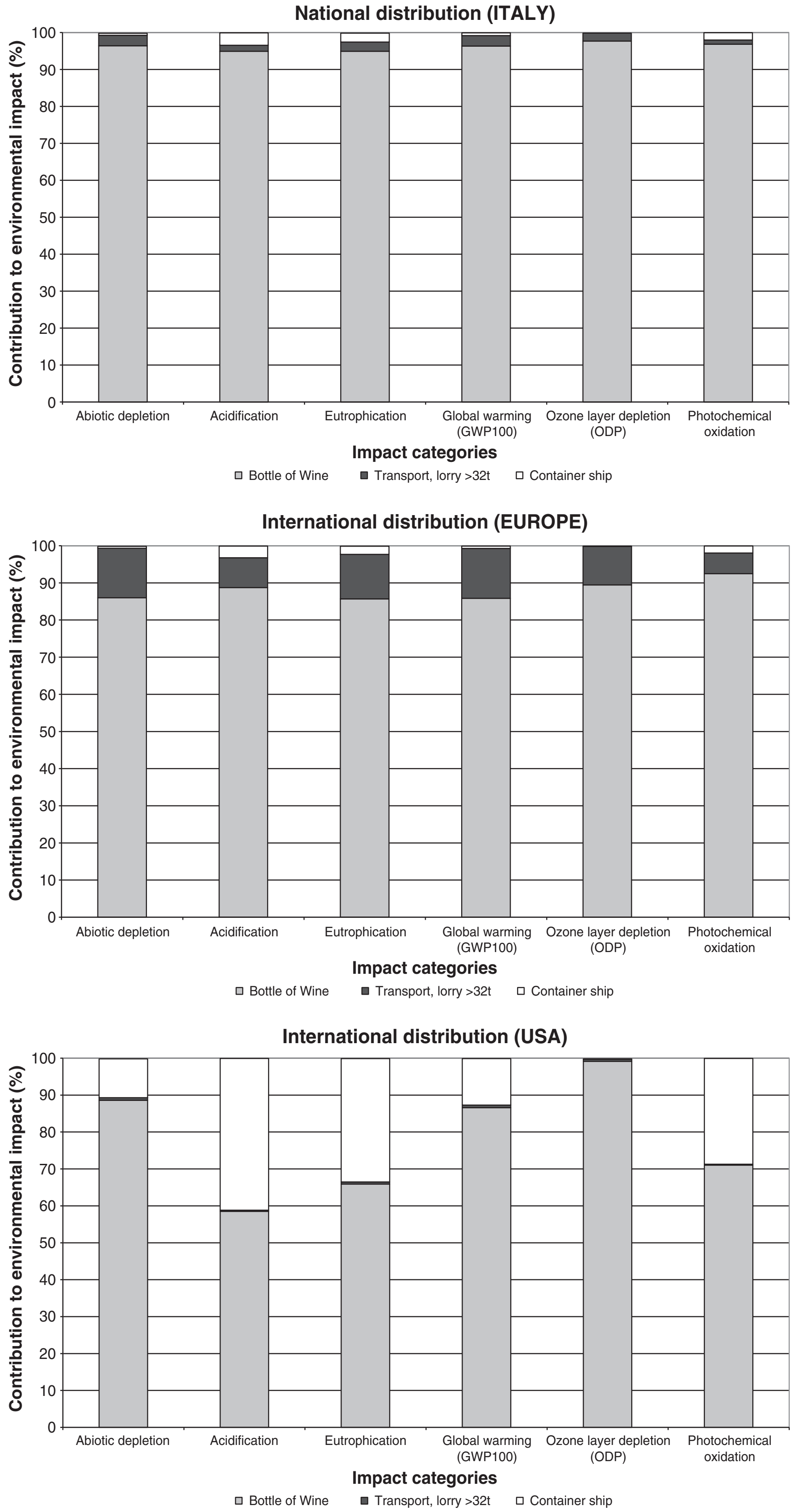


Fig. 5. Results with the inclusion of the transportation phase (percentage contribution to each impact category of the bottle of wine production and transportation phase).

contributions of these sub-phases are shown in Fig. 4. Vine planting had a great influence on the environmental performance of the whole agricultural phase. The main contributor in the vine planting phase to the impact categories considered was the production of diesel fuel needed for field operations (above all: land preparation and trellis laying), which is consistent with the results obtained by Petti et al. (2006), Bosco et al. (2011) (for the GWP category impact) and Benedetto (2013). The contribution of diesel fuel production to environmental impact varied between 85% (abiotic depletion) and 40% (eutrophication).

The agricultural phase is the stage where the largest sensitivities were found. The variation of the sensitivity parameters had large effects on the eutrophication impact category, which was primarily associated with emissions due to the use of fertilisers. Changes in parameters related to the emission of nitrogen compounds resulting from the use of organic and synthetic fertilisers affected eutrophication. Changes in the emission factors for LPG combustion resulted in small changes for global warming. Pesticide emissions were not found to be relevant, regardless of the percentage value used to calculate them. Table 9 shows the results of the sensitivity analysis.

3.2. Level B

The results obtained with the inclusion of the transportation phase are shown in Fig. 5 and in the Supplementary documents (Tables 5, 6 and 7). The impact due to the distribution of one bottle of wine within national borders (Italy) was maximum (5%) for acidification and eutrophication. Data concerning the transportation phase in Italy were provided by the company; therefore, the results were considered reliable.

As for international distribution (Europe, Italy excluded, and USA), the environmental burden associated with transportation increased, as expected. In particular, transportation by sea over long distances (USA) appeared to have a relevant impact on almost all of the impact categories taken into consideration: 41% for acidification, 33% for eutrophication, 29% for photochemical oxidation, 12% for global warming potential and 10% for abiotic depletion. Data concerning international distribution were, however, estimated; therefore, the results obtained have to be considered a rough indication of the role played by the transportation phase on the overall life cycle of a bottle of wine.

The comparison between level A and level B results is shown in Fig. 6. Distribution within national borders caused an increase in environmental impacts of less than 5% with respect to the A scenario for all of the impact categories. European distribution determined a worse environmental performance in every impact category considered, in particular for abiotic depletion (14% increase with respect to level A results), global warming potential (12%) and ozone layer depletion (10%). US distribution was responsible for a consistent increase in the environmental burden, especially for acidification, eutrophication and photochemical oxidation impact categories. It has to be taken into account that the results for US distribution would have been worse if road transportation within the country had also been taken into account.

4. Conclusions

The study carried out evaluates the environmental impacts associated with viticulture, vinification, bottling and packaging in a Sardinian winery. The results showed that the environmental performance of a bottle of Vermentino wine was mostly determined by glass bottle production. Therefore, a reasonable option to reduce the environmental impact of the product would be to use a lighter glass bottle (Aranda et al., 2005; Point et al., 2012; Cleary, 2013) or to substitute the glass bottle with an aseptic carton, although this alternative would require an impact analysis on the chemical and flavour characteristics of the wine (Montedonico, 2005). A study carried out by Pasqualino et al. (2011) showed in fact that, between the glass and aseptic carton options for juice packaging, the second solution had a lower impact on the two environmental categories considered (GWP and Cumulative Energy Demand). The adoption of a lighter container (lighter glass bottle or aseptic carton) would benefit the distribution phase as well; this advantage is proportional to the distance of transportation required. The distribution phase was shown to affect the environmental results as the distance of transportation increased.

The availability of vine planting data allowed us to perform an environmental analysis on the whole agricultural phase involved in the production of grapes. The results showed that the vine planting sub-phase was not negligible given its contribution to the agricultural phase.

The results obtained were compared with other wine-related LCA studies. However, as stated by Neto et al. (2013), the results are not

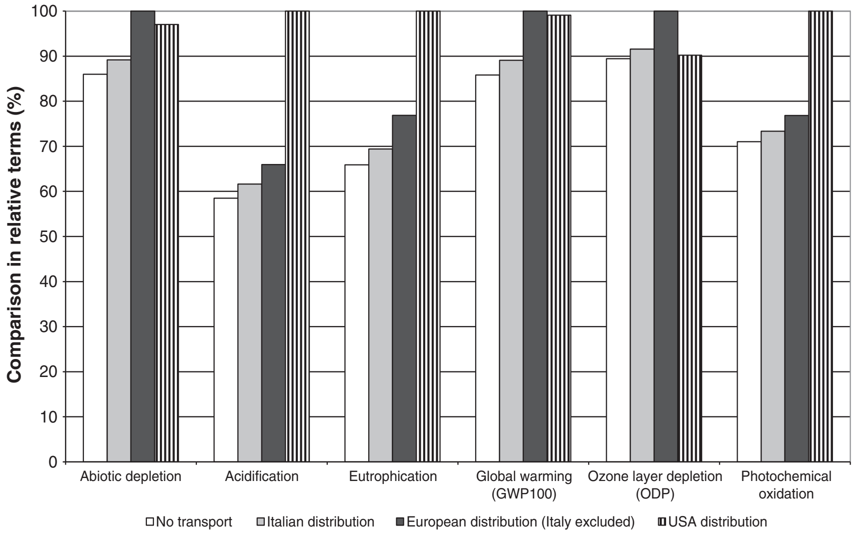


Fig. 6. Comparison between level A and level B results.

easily comparable due to the methodological options available (i.e. the method used to estimate emissions) and the different protocols used to produce the wine. Therefore, harmonised rules allowing a comparison of the results of different studies are needed.

Acknowledgement

The authors wish to thank dott. Stefano Biscaro and the Sella & Mosca for their precious cooperation in providing accurate and reliable data useful in drawing up of the study.

This is the author's contribution: Graziella Benedetto (GB) was responsible for the conception and design. GB had conceived the questionnaire and collected all the data with input from Alessandra Fusi (AF); AF elaborated the data which were interpreted with GB. GB wrote the introduction and results and discussion; AF wrote materials and methods and results and discussion. All authors contributed to writing conclusion. The authors approved the final version.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2013.11.148.

References

Andersson I. Environmental management tools for SMEs: a handbook. Environmental issues series. European Environment Agency; 1998.
Aranda A, Scarpellini S, Zabalza I. Economic and environmental analysis of the wine bottle production in Spain by means of life cycle assessment. Int J Agric Resour Gov Ecol 2005;4:178–91.
Ardente F, Beccali G, Cellura M, Marvuglia A. POEMS: a case study of an Italian wine-producing firm. Environ Manag 2006;3:350–64.
Audsley E. Harmonization of environmental life cycle assessment for agriculture. European Commission DG VI Agriculture. Silsoe: Silsoe Research Institute; 1997.
Autorità Portuale di Genova. http://servizi.porto.genova.it/traffici/statistiche/quota_principali_porti_collegati/dettaglio.aspx?id=11118. [last visit 21/05/2013].
Barber N, Taylor C, Strick S. Wine consumers' environmental knowledge and attitudes: influence on willingness to purchase. Int J Wine Res 2009;1:59–72.
Barry MT. Life cycle assessment and the New Zealand wine industry: a tool to support continuous environmental improvement. Wellington, New Zealand: Massey University; 2011 [Master thesis].
Benedetto G. The environmental impact of a Sardinian wine by partial life cycle assessment. Wine Econ Policy 2013;2:33–41.
Benedetto G, Corinto G, Rugani B, Vázquez-Rowe I, Benetto E. The evolution of life cycle assessment (LCA): some guidelines for future research in the wine sector. Proceedings of the L Annual Meeting of SIDEA, Lecce 26–28 September 2013; 2013.
Berners-Lee M, Howard DC, Moss J, Kaivanto KWA, Scott WA. Greenhouse gas footprinting for small businesses—the use of input–output data. Sci Total Environ 2011;409:883–91.
Bosco S, Di Bene C, Galli M, Remorini D, Massai R, Bonari E. Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy. Ital J Agron 2011;6:93–100.
Burja C, Burja V. Decisions in sustainable viticulture using life cycle assessment. J Environ Prot Ecol 2012;13(3):1570–7.
Carta G. Evaluation of environmental sustainability of two Italian wine productions through the use of the life cycle assessment (LCA) method. Italy: University of Sassari; 2009 [Master thesis, in Italian].
CEC (Commission of the European Communities). Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Integrated product policy. Building on environmental life-cycle thinking. COM(2003)302 final. Brussels, Belgium: CEC; 2003.
Cerutti AK, Bruun S, Beccaro GL, Bounous G. A review of studies applying environmental impact assessment methods on fruit production systems. J Environ Manag 2011;92:2277–86.
CIV—Consorzio Interprovinciale Vini s.c.agr. Environmental product declaration (EPD) and climate declaration–bottled red sparkling wine “Gasparossa Righi”, validated EPD N° S-P-00109, Rev. March 2008, IT. http://www.environdec.com/, 2008.
Cleary J. Life cycle assessment of wine and spirit packaging at the product and the municipal scale: a Toronto, Canada case study. J Clean Prod 2013;44:143–51.
Coderoni S, Bonati G. Impronta carbonica delle aziende agricole italiane. INEA; 2013.
Colman T, Päster P. Red, white, and ‘green’: the cost of greenhouse gas emissions in the global wine trade. J Wine Res 2009;20(1):15–26.
COREVE (Consorzio Recupero Vetro). Programma Specifico di Prevenzione 2010 (Risultati di riciclo 2009); 2010.
CRI (Container Department Institute). http://www.container-recycling.org/. [last visit 29/03/2013].
DETEC (Federal Department of the Environment, Transport, Energy and Communications). http://www.bafu.admin.ch/index.html?lang=it. [last visit 05/02/2013].
Dick J, Smith P, Smith R, Lilly A, Moxey A, Booth J, et al. Calculating farm scale greenhouse gas emissions. Scotland, UK: University of Aberdeen, Carbon Plan, the Macaulay Institute, Pareto Consulting, SAOS Ltd.; 2008.
EMEP/EEA. Agriculture other (use of pesticides and limestone). Air pollutant emission inventory guidebook 2013 —technical report no 12/2013. Copenhagen: European Environment Agency; 2013.
ENVIFOOD Protocol Environmental Assessment of Food and Drink Protocol. European Food Sustainable Consumption & Production Round Table, draft version 0.1, for pilot testing —November 2012; 2012.
FEVE (The European Container Glass Federation). http://www.feve.org/. [last visit 29/03/2013].
Frischknecht R, Jungbluth N, Althaus HJ, Doka G, Heck T, Hellweg S, et al. Overview and methodology. Ecoinvent report no. 1. Dübendorf: Swiss Centre for Life Cycle Inventories; 2007.
FSO (Federal Statistical Office). http://www.bfs.admin.ch/bfs/portal/en/index.html. [last visit 02/05/2013].
Gazulla C, Raugei M, Fullana-i-Palmer P. Taking a life cycle look at Crianza wine production in Spain: where are the bottlenecks? Int J Life Cycle Assess 2010;15:330–7.
Gonzalez A, Klimchuk A, Martin M. Life cycle assessment of wine production process. Finding relevant process efficiency and comparison to eco-wine production, 1N1800 life cycle assessment, group 4; 2006.
González-García S, Castanheira EG, Dias AC, Arroja L. Environmental life cycle assessment of a dairy product: the yoghurt. Int J Life Cycle Assess 2013a;8:796–811.
González-García S, Castanheira EG, Dias AC, Arroja L. Using life cycle assessment methodology to assess UHT milk production in Portugal. Sci Total Environ 2013b;442:225–34.
Guidetti R. L'analisi energetica come indicatore di processo nel settore enologico. In: Blandini G, Manetto G, editors. [s.l]: GeoGeografica, 2005 Jun(Convegno AIIA2005, L'ingegneria agraria per lo sviluppo sostenibile dell'area mediterranea tenutosi a Catania, 27–30 Giugno 2005)88-901860-0-3; 2005.
Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, Koning A, et al. Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: operational annex. III: scientific background. Dordrecht: Kluwer Academic Publishers; 2002 [692 pp.].
Intergovernmental Panel on Climate Change (IPCC). IPCC Guidelines for National Greenhouse Gas Inventories Agriculture, Forestry and Other Land Use, Chapter 11: N_2O emissions from managed soils, and CO_2 emissions from lime and urea application, vol. 4. IPCC; 2006a.
Intergovernmental Panel on Climate Change (IPCC). IPCC Guidelines for National Greenhouse Gas Inventories Energy, Chapter 2: stationary combustion, vol. 2. IPCC; 2006b.
Iribarren D, Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G. Estimation of the carbon footprint of the Galician fishing activity (NW Spain). Sci Total Environ 2010;408:5284–94.
ISO. ISO 14040. Environmental management – life cycle assessment – principles and framework. International Organization for Standardization; 2006a.
ISO. ISO 14044. Environmental management – life cycle assessment – requirements and guidelines. International Organization for Standardization; 2006b.
Lambert G. Carbon neutral wine —Australian winery's ISO 14044 vintage. ISO Focus+, July–August: 34–35; 2010.
Manfredi S, Allacker K, Chomkhamsri K, Pelletier N, Maia de Souza D. Product environmental footprint guide; consolidated version. Deliverable 2 and 4A of the Administrative Arrangement between DG Environment and the Joint Research Centre No N 070307/2009/552517, including Amendment No 1 from December 2010. European Commission (EC), Joint Research Centre (JRC), Institute for Environment and Sustainability (IES). Ref. Ares; 2012 [873782 – 17/07/201].
Margni M, Rossier D, Crettaz P, Jolliet O. Life cycle impact assessment of pesticides on human health and ecosystems. Agric Ecosyst Environ 2002;93:379–92.
Montedonico CM. The life cycle assessment (LCA) of a Chilean cabernet sauvignon wine bottle. (Master thesis)Revista ECOENGEN, Articulo no. 04, 2009, Santiago, Chile. Italy: University of Bologna; 2005.
Neto B, Dias AC, Machado M. Life cycle assessment of the supply chain of a Portuguese wine: from viticulture to distribution. Int J Life Cycle Assess 2013;18:590–602.
Nielsen PH, Nielsen AM, Weidema BP, Dalgaard R, Halberg N. LCA food data base. www.lcafood.dk, 2003.
Notarnicola B, Tassielli G, Nicoletti GM. Life cycle assessment (LCA) of wine production. In: Mattson B, Sonesson U, editors. Environmentally-friendly food processing, 306. Cambridge, England: Woodhead Publishing Ltd.; 2003. p. 326.
Office for National Statistics. http://www.statistics.gov.uk/hub/index.html. [last visit 02/05/2013].
OIV (International Organisation of Vine and Wine). Statistical report on world viticulture 2013; 2013.
Pasqualino J, Meneses M, Castells F. The carbon footprint and energy consumption of beverage packaging selection and disposal. J Food Eng 2011;103:357–65.
Pattara C, Raggi A, Cichelli A. Life cycle assessment and carbon footprint in the wine supply-chain. Environ Manag 2012a;49(6):1247–58.
Pattara C, Cichelli A, Civitarese C, Di Martino M. A comparison of carbon footprints in wine production: the case of two cooperatives wineries in Central Italy, Le Bulletin de l'OIV, vol. 85, no. 977-978-979. ; 2012b. [juillet-aout-septembre].
Petti L, Raggi A, De Camillis C, Matteucci P, Sára B, Pagliuca G. Life cycle approach in an organic wine-making firm: an Italian case-study. Proceedings Fifth Australian Conference on Life Cycle Assessment, Melbourne, Australia, November 22nd to 24th, 2006; 2006. [Available at: http://www.conference.alcas.asn.au/2006/Petti%20et%20al.pdf (last access December 2012)].

Petti L, Ardente F, Bosco S, De Camillis C, Masotti P, Pattara C, et al. State of the art of life cycle assessment (LCA) in the wine industry. International conference on life cycle assessment in the agri-food sector. Bari, Italy, September 22–24 2010; 2010.

Pizzigallo ACI, Granai C, Borsa C. The joint use of LCA and emergy evaluation for the analysis of two Italian wine farms. J Environ Manag 2008;86:396–406.

Point E, Tyedmers P, Naugler C. Life cycle environmental impacts of wine production and consumption in Nova Scotia, Canada. J Clean Prod 2012;27:11–20.

Rossini G, Toscano G, Olivi L. Valorizzazione dei cascami della filiera vinicola. Atti del convegno: Attualità della ricerca nel settore delle energie rinnovabili da biomassa. 16–17 Dicembre 2010, Ancona; 2010.

Rugani B, Niccolucci V, Pulselli RM, Tiezzi E. A cradle-to-gate life cycle assessment integrated with emergy evaluation: sustainability analysis of an organic wine production. Proceedings of the SETAC Europe 19th Annual Meeting on Protecting Ecosystem Health: Facing the Challenge of a Globally Changing Environment, May 31st–June 4th, 2009, Göteborg, Sweden; 2009. p. 274.

Rugani B, Vázquez-Rowe I, Benedetto G, Benetto E. A comprehensive review of carbon footprint analysis as an extended environmental indicator in the wine sector. J Clean Prod 2013;54:61–77. [1 September].

Saxe H. LCA-based comparison of the climate footprint of beer vs. wine & spirits. Institute of Food and Resource Economics, Report no. 207, (on-line, LCA-based comparison of the climate footprint of beer vs. wine & spirits)978-87-92591-02-9; 2010.

Schlich EH. From vineyard to point of sale: allocation of energy use and CO_2-emission to entire supply chains of wine. Proceedings of the 4th Annual Conference, American Association of Wine Economists, June 25–28, 2010, University of California, Davis, California USA; 2010.

Smil V. Phosphorus in the environment: natural flows and human interferences. Annu Rev Energy Environ 2000;25:53–88.

Smith P, Martino D, Cai Z, Gwary D, Janzen H, Kumar P, et al. Agriculture. In climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In: Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA, editors. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press; 2007.

Statistical Offices of the Länder. http://www.statistikportal.de/Statistik-Portal/en/. [last visit 02/05/2013].

United States Environmental Protection Agency (USEPA). Emission factor documentation for AP-42, section 9.12.2, wines and brandy, final report; 1995.

Vázquez-Rowe I, Villanueva-Rey P, Moreira MT, Feijoo G. Environmental analysis of Ribeiro wine from a timeline perspective: harvest year matters when reporting environmental impacts. J Environ Manag 2012;98:73–83.

Vázquez-Rowe I, Rugani B, Benetto E. Tapping carbon footprint variations in the European wine sector. J Clean Prod 2013;43:146–55.

Villanueva-Rey P, Vázquez-Rowe I, Moreira MT, Feijoo G. Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. J Clean Prod 2013. http://dx.doi.org/10.1016/j.jclepro.2013.08.026.

Zabalza I, Aranda A, Scarpellini S. Analysis and improvement of energy and environmental costs for small and medium enterprises in the wine sector. Proceedings of the 16th International Conference on Efficiency, Costs, Optimization, Simulation and Environmental Impact of Energy Systems (ECOS 2003), June 30th–July 2nd, 2003, Copenhagen, Denmark; 2003.

Zamagni A, Masoni P, Buttol P, Raggi A, Buonamici R. Finding life cycle assessment research direction with the aid of meta-analysis. J Ind Ecol 2012;16:39–52.

Science of the Total Environment 494–495 (2014) 119–128

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Environmental profile of paddy rice cultivation with different straw management

Alessandra Fusi [a,1], Jacopo Bacenetti [a,*,1], Sara González-García [b], Annamaria Vercesi [a], Stefano Bocchi [a], Marco Fiala [a]

[a] Department of Agricultural and Environmental Sciences, Production, Landscape, Agroenergy, Università degli Studi di Milano, Via G. Celoria 2, 20133 Milan, Italy
[b] Department of Chemical Engineering, School of Engineering, University of Santiago de Compostela, 15782 Santiago de Compostela, Spain

HIGHLIGHTS

- Italy is the most important European country in terms of rice production (paddy rice).
- The Life Cycle Assessment method was applied from a cradle-to-field gate perspective.
- The environmental profile of rice was analyzed for 7 different impact categories.
- Environmental impact is mainly due: field emissions, fuel consumption and the drying.
- Collection and sale of the straw can improve the environmental performance of rice.

ARTICLE INFO

Article history:
Received 28 February 2014
Received in revised form 24 June 2014
Accepted 27 June 2014
Available online xxxx

Editor: E. Capri

Keywords:
Life Cycle Assessment
Straw management
Allocation
Environmental performance
Methane emission

ABSTRACT

Italy is the most important European country in terms of paddy rice production. North Italian districts such as Vercelli, Pavia, Novara, and Milano are known as some of the world's most advanced rice cultivation sites. In 2013 Italian rice cultivation represented about 50% of all European rice production by area, and paddy fields extended for over 216,000 ha. Cultivation of rice involves different agricultural activities which have environmental impacts mainly due to fossil fuels and agrochemical requirements as well as the methane emission associated with the fermentation of organic material in the flooded rice fields.
In order to assess the environmental consequences of rice production in the District of Vercelli, the cultivation practices most frequently carried out were inventoried and evaluated.
The general approach of this study was not only to gather the inventory data for rice production and quantify their environmental impacts, but also to identify the key environmental factors where special attention must be paid. Life Cycle Assessment methodology was applied in this study from a cradle-to-farm gate perspective.
The environmental profile was analyzed in terms of seven different impact categories: climate change, ozone depletion, human toxicity, terrestrial acidification, freshwater eutrophication, marine eutrophication, and fossil depletion. Regarding straw management, two different scenarios (burial into the soil of the straw versus harvesting) were compared.
The analysis showed that the environmental impact was mainly due to field emissions, the fuel consumption needed for the mechanization of field operations, and the drying of the paddy rice. The comparison between the two scenarios highlighted that the collection of the straw improves the environmental performance of rice production except that for freshwater eutrophication.
To improve the environmental performance of rice production, solutions to save fossil fuel and reduce the emissions from fertilizers (leaching, volatilization) as well as methane emissions should be implemented.

1. Introduction

The food sector, including agricultural subsystems, transport, processing, and disposal, is responsible of a remarkable environmental impact (IPCC, 2006; FAO, 2013a). This impact is related with the use of energy, agrochemicals, fossil fuels, and machinery in agricultural activities (Glithero et al., 2011; Nalley et al., 2011; Gonzalez Garcia at al., 2012).

In the future, agriculture will face a challenge to increase food production and simultaneously reduce environmental impact. The first condition to meet this impressive challenge is the evaluation of the environmental impact of the different agricultural activities, and then

* Corresponding author.
E-mail address: jacopo.bacenetti@unimi.it (J. Bacenetti).
[1] These authors equally contributed to this study.

http://dx.doi.org/10.1016/j.scitotenv.2014.06.126
0048-9697/

the identification of options available to reduce it (Brentrup et al., 2004; Leip and Bocchi, 2007).

Over the years, the evaluation of agricultural activities has moved from economic and energetic aspects to environmental ones (Fiala and Bacenetti, 2012). Today, the concern for environmental performance is continuously increasing (Bessou et al., 2013). For this reason, an accepted scientific basis for impact assessment is required for transparent and credible communication with the general public (Buratti and Fantozzi, 2010).

In the last decade, several studies have been conducted to evaluate the environmental impact of agricultural systems (Nemecek et al., 2011; Grab et al., 2013). Different types of agricultural products (cereals, fiber crops, potato, fruits) have been analyzed, also taking into account different production systems (Romani and Beltarre, 2007; Hokazono and Hayashi, 2012; Bacenetti et al., 2014). Large differences can be found in the environmental results mainly due to differences in management practices (Weiss and Liep, 2012; Roer et al., 2012; Negri et al., 2014) as well as climatic and soil conditions (Goglio et al., 2012; Hokazono and Hayashi, 2012).

Rice is one of the most important agricultural commodities in the world. It is cultivated mainly for human food. According to the FAO forecast for global paddy production, it has been set at 46.4 million tons (497.6 million tons, milled basis) in 2013 with a global area of 163.5 million hectares (FAO, 2013b). Rice is, by far, the most important food in many regions, both tropical (India, Indonesia) and temperate (China, Japan) (FAO, 2013a,b). Rice cultivation not only generates wealth and jobs for the cultivating regions but also causes remarkable environmental impacts from their production practices (Leip, 2007; Blengini and Busto, 2009; Eshun et al., 2013; Yoo et al., 2013). Apart from soil and water pollution and consumption of energy and production factors, paddy fields (irrigated or flooded) are claimed to be responsible for 10–13% of worldwide methane anthropogenic emissions (Wang et al., 1997). Thus, rice cultivation contributes to a great extent of the global warming phenomenon (Roy et al., 2007, 2009a,b).

Although far from the main world producers, Italy is the most important European country in terms of rice production. Paddy fields extended over 216,019 ha in 2013 (Ente Risi, 2013). Italian rice fields represent about 50% of European rice dedicated fields (Blengini and Busto, 2009; Ente Risi, 2013). Italian rice production in 2013 was 1.4 million tons, about 14% lower than the rice yield in 2012. North Italian districts (such as Vercelli, Pavia, Novara, and Milano) are known as some of the most advanced rice cultivation areas in the world. These four districts produce more than 85% of Italian rice (Ente Risi, 2013).

As previously mentioned, agricultural practices have environmental impacts which are related with agrochemical use and fossil fuel consumption in agricultural equipment. However, unlike others crops, high methane emissions are generated when rice is cultivated in flooded paddy fields due to the anaerobic decomposition of organic matter (IPCC, 2006; Kanta Gaihre et al., 2014). For this reason, a detailed identification and assessment of rice production activities represent the first step for the reduction of rice-derived environmental impacts.

Life Cycle Assessment (LCA) is a standardized methodology designed for the evaluation of the potential environmental impacts of products (processes or services) throughout their whole life cycle (ISO, 2006). Originally developed for industrial systems, LCA is becoming more important in the agro-food sector (Roy et al., 2009a,b). Today, LCA is also accepted and used for the evaluation of agricultural activities, where it can be applied to: i) detect the environmental hotspots (processes or activities responsible for the main share of the environmental impacts) and, ii) to compare different processes or different technical solutions that can be implemented in the same process (Harada et al., 2007; Blengini and Busto, 2009; Hatcho et al., 2012; Bacenetti et al., 2013).

Rice cultivation in Northern Italy can be carried out following several management practices. In particular, the main differences can be found regarding flooding systems, fertilizations, and straw management (Baldoni and Giardini, 2000; Leip and Bocchi, 2007). These differences are the result of social, economic, and climatic aspects (FAO, 2013b). A definition of a "standard cultivation practice" (SCP) can be useful in order to compare different rice production practices and to understand the impact on the environment among several alternative solutions. For each rice cultivation district, the standard cultivation practice represents the cultivation technique most frequently used.

In this paper, the environmental impact derived from rice cultivation in the Vercelli district under SCP has been evaluated using the LCA methodology. The Vercelli district is the most important Italian region in terms of rice production, representing approximately 33% of the total national yield (Blengini and Busto, 2009). This study has been performed from a cradle-to-farm gate perspective, taking into account detailed information (rice yield and field operations) for the years 2009–2013. In addition, the environmental hotspots have been identified throughout the rice production system. An alternative scenario focused on straw management has been proposed and environmentally evaluated in order to identify environmental differences and improvements.

2. Materials and methods

2.1. Goal and scope definition

The goal of this study is the evaluation of the environmental performance of rice cultivation in the Vercelli district following the SCP. Moreover, the most critical agricultural processes for the rice system under study throughout its life cycle were identified in order to propose improvement alternatives.

Rice is one of the most widespread cereal crops in the Lombardy and Piedmont regions. Rice is grown mainly in the eastern part of the Po Valley area. The Vercelli district (Piedmont) is the most important Italian area for rice cultivation (around 33% of total Italian rice fields) (45°19′00″N, 8°25′00″E). In this area, cultivation is characterized by good water availability and restricted manure and slurry availability due to limited livestock activities. Therefore, in these climatic conditions, rice fields are managed in a similar way and a standard cultivation practice (SCP) is predominant compared to other alternative practices. For this reason the SCP was evaluated using the LCA methodology. As mentioned by Zamagni et al. (2012), the European Commission states that "LCAs provide the best framework for assessing the potential environmental impacts of products currently available" (CEC, 2003). Life Cycle Assessment (LCA) is a standardized methodology used for estimating the environmental burdens associated with life cycle of products or processes (ISO, 2006). This methodology is considered to be effective for evaluating environmental performance in the agro-food and beverage sector (Roy et al., 2009a,b).

2.2. Functional unit

According to ISO standards, the functional unit (FU) is defined as a quantified performance of a product system to be used as a reference unit in an LCA (ISO, 14040, 2006). The functional units (FU) most frequently selected in LCA studies of agricultural production systems are:

(1) the mass of product (grain, fruit, biomass, milk, etc.) (Nalley et al., 2011; Gan et al., 2011; González-García et al., 2012; Murphy and Kendall, 2013);
(2) the cultivated area (e.g., 1 ha) (Cellura et al., 2012; Negri et al., 2014).

In this study, in view of the comparison between the environmental burdens of SPC and the ones of alternative cultivation practices (potentially with different yields), 1 ton of paddy rice (commercial moisture content 14%) has been chosen as FU.

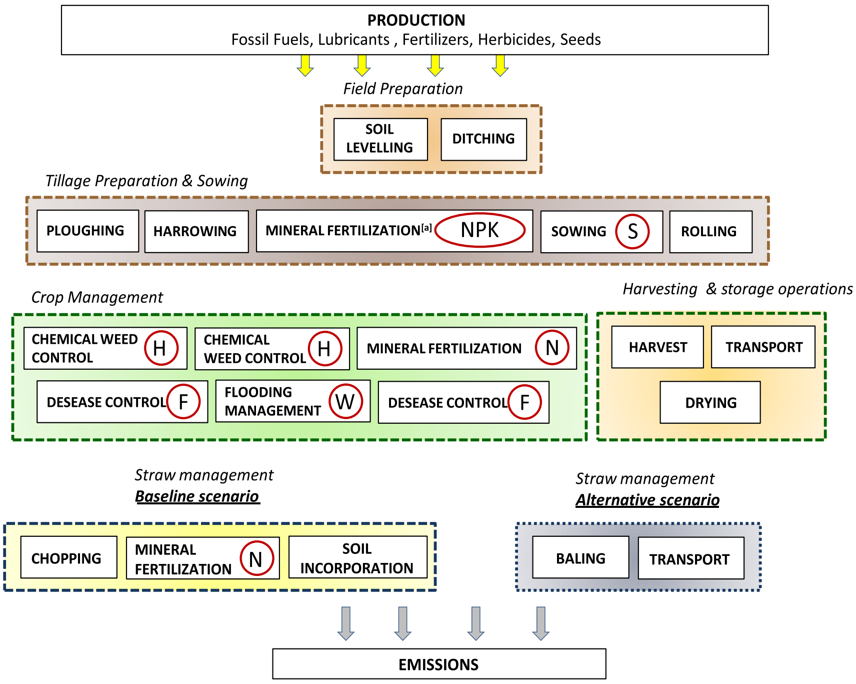


Fig. 1. System boundaries.

2.3. Description of rice standard cultivation practice (SCP)

Rice (*Oryza sativa* spp. L.) is a summer crop in temperate regions where different varieties are grown (Enterisi, 2013). In the Vercelli district of Northern Italy, the local climate is characterized by an average annual temperature of 12.7 °C and rainfall is mainly concentrated in autumn and spring (average annual precipitation is equal to 745 mm). The most widespread varieties in the District of Vercelli are characterized by a long-grain paddy (CL26, Gladio and Dardo) and are mainly cultivated in flooded paddy fields.

This study has been carried out from a cradle-to-farm gate perspective. In more detail, for the present analysis, the LCA model was carried out by including five subsystems: (1) soil preparation, (2) soil tillage

Table 1
Processes involved in SCP.

Subsystem	Field operation	Operative machine	Tractor		Fuel consumption	Input		Time
			kW	kg	kg·ha^{-1}	Product	Amount (ha^{-1})	h/ha
Soil preparation	Leveling	Laser level	140	5850	19.2			3.20
	Ditching	Trenchers	100	5600	8.5			0.70
Soil tillage & seeding	Ploughing	Plough[a]	135	7600	27.7			1.10
	Harrowing	Rotary harrow	90	5050	18.6			1.70
	Mineral fertilization	Fertilizer spreader	90	5050	3.1	N P_2O_5 K_2O	65 kg urea 50 kg as superphosphate 45 as KCl	0.20
	Seeding	Seeder	90	5050	8.4	Seeds	200 kg	0.20
Crop management	Mineral fertilization	Fertilizer spreader	90	5050	3.1	N P_2O_5 K_2O	70 kg urea 0 0	0.25
	Weed control pre seeding	Sprayer	90	5050	3.3	Herbicide	4.5 dm^3 pendimethalin (a.s.[b] = 30.7%)	0.20
	Weed control post germination	Sprayer	90	5050	3.3	Herbicide	0.875 dm^3 imazamox (a.s. = 3.7%) diazolecomp	0.20
	Weed control post germination	Sprayer	90	5050	3.3	Herbicide	0.875 dm^3 imazamox (a.s. = 3.7%) diazolecomp	0.20
	Disease control	Sprayer	90	5050	3.5	Fungicide	0.3 dm^3 tricyclazole (a.s. = 75%)	0.20
	Water management	–			–	Water	25,000 m^3	–
	Disease control	Sprayer	90	5050	3.5	Fungicide	0.3 dm^3 tricyclazole (a.s. = 75%)	0.20
Harvesting & storage	Harvest	Combine harvester	335	15,500	36.1			0.80
	Transport	Trailer	90	5050	15.1			0.80
	Transport	Trailer	90	5050	15.1			0.80
	Drying	Dryer						–
Straw management	Straw chopping	Chopper	90	5050	18.5			1.00
	Mineral fertilization	Fertilizer spreader	90	5050	3.3	N	40 kg urea	0.20
	Soil incorporation	Disk plough			26.1			1.00

[a] Depth = 35 cm.
[b] a.s = active substance.

and seeding, (3) crop growth, (4) harvesting, transport and storage, and (5) straw management.

System boundaries are shown in Fig. 1. Table 1 reports detailed data about rice cultivation processes.

The two general guiding principles of the SCP in the District of Vercelli are:

(1) Fertilization based on the nutrient removal by crop (Baldoni and Giardini, 2000) without extra nutrient application. According to the maintenance concept, only nutrients that have been removed with the crop (grain and straw) at harvest process are returned with the fertilization. If straw is not collected, the nutrient removal is computed considering only the paddy rice production.
(2) Absence of red rice (*O. sativa* L. var. *sylvatica*). Thus, there is neither a chemical weed control before sowing nor a false seeding technique.

The field operations are divided into five main subsystems:

Subsystem 1: soil preparation Compared with other cereals, when the rice is cultivated in a flooded paddy, the seed-bed preparation involves specific operations that are carried out before the tillage operations. To obtain proper water management during the rice growing season, the paddy is leveled and some furrows are dug. The leveling and the ditching are operations carried out to prepare the soil for the rice cultivation in flooded conditions. Their intensity is quite variable and depends on the soil conditions (i.e., texture, slope), climate, tillage operations, and water management. However, the ditching operation is carried out every year while the leveling, mainly due to high fuel consumption and the requirement of operative machines (laser levels) not usually present in the farm machinery fleet, is done only when needed. In the SCP, the leveling is carried out every three years. In other rice districts (for example the Ferrara rice district located in the delta of the Po River) with clay soils and low slope, the leveling operation is carried out more rarely.

Subsystem 2: soil tillage and seeding The soil tillage operations include a ploughing (average 30 cm deep) as primary tillage and by a harrowing as secondary tillage. After the ploughing, the mineral fertilization is carried out by a fertilizer spreader. Thus, the amount of phosphorous (50 $kg \cdot ha^{-1}$ as superphosphate) and potassium (45 $kg \cdot ha^{-1}$ as potassium chloride) is spread together with the nitrogen fertilizer (70 $kg \cdot ha^{-1}$ as urea). The sowing is performed in non-flooded fields (dry conditions) using a seeder and about 200 $kg \cdot ha^{-1}$ of rice seed (paddy rice) in order to obtain a crop density of 300–350 $plants \cdot m^{-2}$. After sowing, in order to help the germinability of the seeds, a rolling is carried out.

Subsystem 3: crop growth This subsystem involves four steps: (1) the chemical weed control carried out twice, one in pre-emergence and one in post-emergence; (2) top fertilization with urea application (65 $kg \cdot ha^{-1}$); (3) rice blast (*Pyricularia oryzae* [Cavara]) control by means of two fungicide applications (0.3 $kg \cdot ha^{-1}$ of Beam, containing the active ingredient tricyclazole). The first at cum elongation and the second at panicle initiation-booting; (4) water management with a global water consumption of 25,000 $m^3 \cdot ha^{-1}$ and two aeration periods during the cropping season (the first in middle June for carried out the chemical weed control and the second at the end of June for the top fertilization). Water is an important tool to protect rice from cold that can cause yield losses due to spikelet sterility. The water is removed from the field when the kernel reaches the waxy ripeness (approximately 2 weeks before the harvest).

Subsystem 4: harvesting, transport, and storage The harvesting operations are carried out by combine harvester when the moisture content of rice grain is 20–30% (depending on climatic conditions). The rice paddy is loaded into farm trailers coupled with tractors, and then it is transported to the farm where the paddy rice is dried to a humidity of 14% by means of a farm dryer. The straw is left in the soil.

Subsystem 5: straw management The straw management involves the chopping of the biomass left in the soil and its burial into the soil after an application of urea (40 $kg \cdot ha^{-1}$). Considering the high C/N ratio of the straw (around 50–70) this supply of nitrogen is important to help the decomposition of the straw.

2.4. System boundaries

This study applies a cradle-to-farm gate perspective, including in the analysis the following steps of rice production:

1) Crop cultivation and harvesting
2) Transport to the farm
3) Drying

The system boundaries are reported in Fig. 1.

The lifecycle of each agricultural process was included within the system boundaries: raw material extraction (e.g., fossil fuels and minerals), manufacture (e.g., seeds, fertilizers and agricultural machines), use (diesel fuel consumption and derived combustion and tire abrasion emissions), maintenance and final disposal of machines, and supply of inputs to the farm (e.g., fertilizers and herbicides). The indirect environmental burdens of capital goods where included because of the high level of mechanization of the Vercelli farms. For the same reasons, capital goods relevant to the post-harvest phase (drying and storing) were also included.

Table 1 reports the agronomic inputs for the system under assessment, as well as the characteristics of the agricultural machines commonly used.

Regarding the straw management, two different scenarios were considered taking into account that they are the current practices carried out in Italy:

1. The baseline scenario (BS): This is the most widespread situation and is described in Table 1. After being chopped, the burial of the straw into the soil was considered.
2. The alternative scenario (AS): The straw is collected by bailing and then the bales are sold. In this scenario, the applied amounts of mineral fertilizers are higher because the straw collection increases the removal of nutrients. Also the methane emissions from the soil have been recomputed taking into consideration that the straw is not buried into the soil and therefore less organic matter is decomposed in anaerobic conditions.

Between these two straw management scenarios, the solution most usually carried out is the straw collection. The straw burial into the soil is performed in areas where the demand for this byproduct is low.

2.5. Inventory analysis

The activities performed in the SCP-based system under study were identified by means of interviews with experts (academic professors, big farmers, and technicians), as well as with surveys of farms in the

Table 2
Rice cultivation in flooded paddy field: nutrient removal by the crop (kg of nutrient for tons of rice paddy).

Nutrient	Unit	Total	Rice Paddy	Straw
N	$kg \cdot t^{-1}$	17.5	11.0	6.5
P_2O_5	$kg \cdot t^{-1}$	10.5	7.0	3.0
K_2O	$kg \cdot t^{-1}$	25.5	5.5	20.5

Vercelli District, which were chosen with the help of Ente Nazionale Risi (www.enterisi.it) (Enterisi, 2012), a public corporation involved in assisting the operators of the whole rice chain.

Data concerning field operations and drying were also obtained via questionnaires that were distributed to farmers. More specifically, information regarding fertilizers and pesticides was also collected by consulting the "Quaderni di campagna," a mandatory document in which their use must be reported. The diesel fuel consumption was estimated by using the model SE^3A (Fiala and Bacenetti, 2012) that considers the power requirements of the operative machines and their work capacities. For all the field operations, a proper coupling among tractors and implements was considered.

Regarding fertilization, the information about nutrient (N, P_2O_5, and K_2O) removal from crops (Table 2) was taken from Baldoni and Giardini (2000).

Emissions due to the fertilizer applications were also included within the system boundaries. Nitrogen emissions (nitrate, ammonia, and nitrous oxide) were computed following the IPCC Guidelines (2006); while phosphate emissions were calculated in accordance with Smil (2000) (losses of P equal 1% of the total applied phosphorus by means of fertilizers and crop residues).

Methane emissions from anaerobic decomposition were computed in accordance with the IPCC methodology (IPCC, 2006). More specifically, the default methane emission factor suitable for the case of no flooding for less than 180 days before rice cultivation and then continuous flooding during rice cultivation without organic amendments (Neue et al., 1996; Wang et al., 1997; IPCC, 2006) was considered. This emission factor (1.30 kg $CH_4 \cdot ha^{-1} \cdot day^{-1}$) was corrected with the scaling factors for: (1) water regime before and during cultivation, (2) number of aeration periods, and (3) application of organic matter (IPCC, 2006). In more detail, (1) two aerations, (2) a non-flooded preseason longer than 180 days, and (3) a long straw incorporation before the cultivation (more than 30 days) were considered.

Pesticide derived emissions were also estimated in accordance with Margni et al. (2002) and Althaus et al. (2007). According to these studies, the fraction of active substances entering the soil is assumed to be 85% of the total mass applied quantity. In other words, around 5% of the pesticide rate stays on the plant and 10% is emitted into the air. The run-off of the active ingredients from the soil into the water is assumed to be 10% maximum of the pesticide rate.

Background data for the production of rice seeds, diesel fuel, fertilizers, and pesticides were obtained from the Ecoinvent database (Althaus et al., 2007; Frischknecht et al., 2007; Jungbluth et al., 2007; Nemecek and Käggi, 2007; Spielmann et al., 2007).

Fields under study were previously dedicated to rice cultivation. Therefore, no change in the overall soil carbon content was assumed (Gonzalez-Garcia et al., 2012; Gonzalez-Garcia et al., 2013; Gan et al., 2011).

Information concerning the paddy rice average yield was taken from the annual report of Ente Nazionale Risi (Enterisi, 2013). Data about the prices of paddy rice (average, max, and min value) and straw came from the price list of the Chamber of Commerce of the Vercelli District (2013). Production of straw was computed considering the Harvest Index (HI). The HI expresses the dry mass of a harvested product (the paddy rice) as a percentage of the total crop dry mass. Knowing the HI, the mass of straw can be calculated proportionally. In this study, an HI equal to 39% was employed; this value was detected by Boschetti et al. (2006) for the cultivation of the same rice varieties in Northern Italy.

Table 3
Inventory data for rice cultivation and drying (data refer to the FU).

	Unit	BS	AS
Input			
Urea (1st fertilization)	kg	4.60	8.85
Superphosphate (1st fertilization)	kg	7.69	10.77
Potassium chloride (1st fertilization)	kg	6.92	25.38
Rice seed	kg	30.77	30.77
Pendimethalin (1st application)	kg	0.21	0.21
Diazole-compounds (2nd application)	kg	0.005	0.005
Diazole-compounds (3rd application)	kg	0.005	0.005
Urea (2nd fertilization)	kg	4.95	8.85
Potassium chloride (2nd fertilization)	kg	6.92	–
Beam (4th application)	kg	0.04	0.04
Beam (5th application)	kg	0.04	0.04
Urea (3rd fertilization)	kg	2.83	2.83
Diesel fuel consumption	kg	33.07	33.41
Water consumption	m^3	3846	3846
Transport by tractor	km	2.0	2.0
Fuel oil for grain drying	MJ	580.31	580.31
Electricity for grain drying	kWh	176.92	176.92
Output			
Rice (14% moisture)	t	1.00	1.00
Straw (dry matter)	t	1.35	1.35
Emissions to air			
Dinitrogen monoxide	kg	0.56	0.37
Ammonia	kg	4.76	2.15
Methane	kg	15.34	7.28
Emissions to water			
Nitrate	kg	34.22	23.51
Phosphate	kg	0.72	0.16

Table 3 reports the data inventory for rice production, both for BS and AS. Data refer to the FU.

2.6. Allocation

During the rice cultivation process, another product, straw, is produced. Straw is usually chopped and incorporated into the soil to improve quality, but it could also be used as animal bedding or for energy generation.

In this study, the straw incorporation into the soil has been considered as baseline scenario. Thus, no allocation was carried out. All the environmental burdens associated with rice cultivation were attributed to the rice grains.

Nonetheless, in order to provide a fuller overview of the available uses of straw, an economic-based allocation method has been proposed. This allocation method has been chosen since the market prices reflect the relevance of the different co-products obtained through rice cultivation. Moreover, economic allocation has been used in other related studies focused on food products in order to solve the problem of multifunctionality (Andersson, 2000; Blengini and Busto, 2009; Fallahpour et al., 2012; Fusi et al., 2013).

Table 4 reports the dry matter yields, the market prices (average value), and the allocation factors considered for each co-product.

Table 4
Parameters for economic allocation.

Product	Yield[a] ($t \cdot ha^{-1}$)	Price[b] ($€ \cdot t^{-1}$)	Allocation factors (%)	
			Baseline scenario	Alternative scenario
Rice Paddy	5.59	320[c]	100	82.6
Straw	8.74	50	0	17.4

[a] Dry matter
[b] Average price during year 2013 at the Chamber of Commerce of Vercelli District.
[c] 14% moisture content (5.59 t of dry matter = 6.50 t at 86% of dry matter).

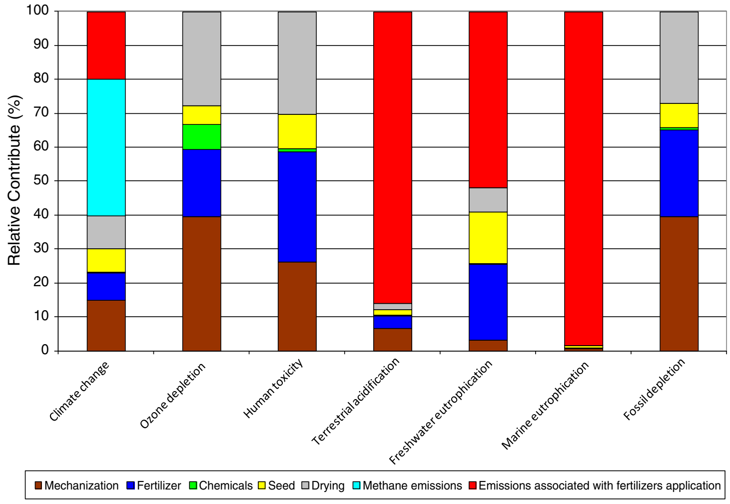


Fig. 2. Environmental impact for the baseline scenario (BS).

2.7. Impact assessment

Among the steps defined within the lifecycle impact assessment (LCIA) stage of the standardized LCA methodology, the classification and characterization stages were carried out in this study. The characterization factors at midpoint level reported by the ReCipe method were used (Goedkoop et al., 2009). According to other studies related with agricultural systems (Gonzalez-Garcia et al., 2012; Bacenetti et al., 2014; Cellura et al., 2012), the following impact categories were selected to evaluate the environmental profile of rice cultivation system: climate change (CC), ozone depletion (OD), human toxicity (HT), terrestrial acidification (TA), freshwater eutrophication (FE), marine eutrophication (ME), and fossil depletion (FD). The SimaPro 8.0.1 software was used for the computational implementation of the lifecycle inventories.

2.8. Sensitivity analysis

In order to test the robustness of the results and to investigate the influence of the choices made in the modeling phase, a sensitivity analysis was carried out on the system under study. Due to their relevance on the environmental impact, the parameters associated with the emission of methane were considered. The emission factors for methane emissions were modified within the range defined by IPCC (minimum and maximum values). The analysis was carried out considering, first, all minimum values of emission factors and, second, all maximum values. The sensitivity analysis was performed on the BS scenario.

Moreover, a sensitivity test was performed in order to gauge the reliability of the economic allocation adopted for the AS. Instead of the average values, the maximum and minimum prices of rice and straw were utilized. In 2013, the minimum and maximum prices were 305–371 €/t and 40–75 €/t, respectively, for rice and straw.

3. Results

3.1. General environmental results

Fig. 2 reports the results corresponding to the environmental impact assessment for the SCP-based system. In the figure, the impacts of mechanization were gathered along with the ones for fertilizers and

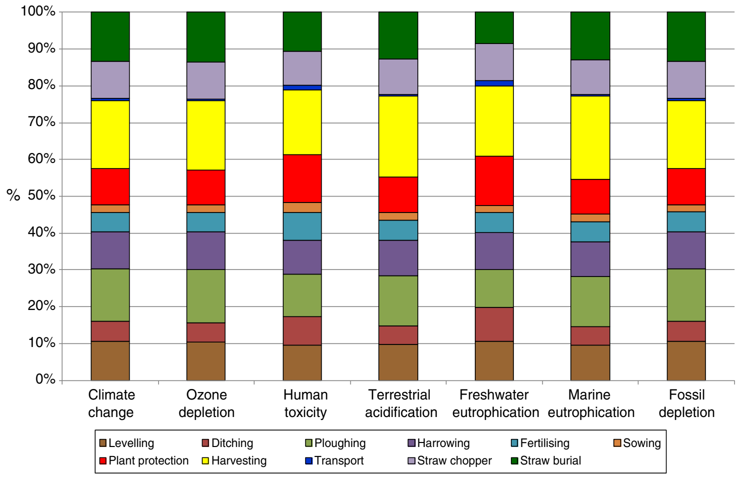


Fig. 3. Impact of field mechanization: Relative contribution of the different field operations.

Table 5
Comparison between the two scenarios.

Impact category		Baseline score	Alternative score
Climate change	CC	669.946 kg CO_2 eq	416.182 kg CO_2 eq
Ozone depletion	OD	0.045 g CFC-11 eq	0.039 g CFC-11 eq
Human toxicity	HT	22.543 kg 1.4-DB eq	20.749 kg 1.4-DB eq
Terrestrial acidification	TA	13.870 kg SO_2 eq	6.148 kg SO_2 eq
Freshwater eutrophication	FE	70.547 g P eq	75.501 g P eq
Marine eutrophication	ME	8.470 kg N eq	4.758 kg N eq
Fossil depletion	FD	108.469 kg oil eq	94.302 kg oil eq

agro-chemical productions and emissions, in air (N_2O, NH_3) and in water (PO_4^{3-}, NO_3^-), associated with fertilizer applications.

The emissions associated with fertilizer applications (leaching, nitrous oxide production, and volatilization) had a deep impact over all the impact categories and, in particular, over TA (86.0%) and eutrophication (FE 51.8% and ME 98.3%). Methane emissions from the soil, due to the anaerobic decomposition of organic matter, are by far the main emission source for CC (40.4%). Field mechanization (that groups emissions related to diesel fuel production and consumption with manufactory of tractors and implements) plays a key role on FD (39.6%), OD (39.5%), and HT (26.2%). The production of mineral fertilizers is important for HT (32.4%), FE (22.3%), FD (25.4%), and OD (20.0%). Finally, the production of agro-chemicals (herbicides and fungicides) has little impact (less than 1.5%) in all the assessed impact categories except than for OD (7%), while the drying is important for HD (30.3%), OD (27.7%), and FD (27.1%).

As mentioned in Section 2.8, a sensitivity analysis was performed on the BS scenario. The variations of the IPCC emission factors have large effects on the CC impact category, which is primarily associated with emissions of methane. The variation of CC results produced by changing the IPCC emission factors ranges from −26% to +77.2%.

In Fig. 3, the impact of field mechanization is shown subdividing among the different operations. The different field operations have a quite stable incidence on the assessed impact categories. Considering that among these emission sources, the production and consumption of diesel fuel represent the two major contributors to the environmental burden of the mechanization, the field operations having the higher impact are the harvesting (about 18%) and the ploughing (about 13%). These operations are characterized by high power requirements.

3.2. Alternative scenario environmental profile

Table 5 reports the results of the environmental impact assessment for the two scenarios considered: baseline (BS, straw incorporated into the soil) and alternative (AS, straw baled and sold).

Fig. 4 reports the relative contribution of the inputs and outputs for the AS: The main difference with respect BS concerns the fertilizers' production, which in the AS affects the human toxicity category at 42.3%, ozone depletion at 28.5%, freshwater eutrophication at 24.5%, and climate change at 19.7%.

The comparison between the baseline and alternative scenarios is shown in Fig. 5. The collection and selling of the straw (AS) allowed a reduction of the environmental burden for all the impact categories evaluated (from −8% for human toxicity to −56% for terrestrial acidification), except for freshwater eutrophication (+7%). In the AS, a higher amount of mineral fertilizer was applied in order to balance the higher nutrient removal and, consequently, the N leaching increased as did the emissions for nitrogen volatilization and denitrification. Regarding the climate change, the differences are mainly due to the reduction of methane emissions from soil that are lower (99.70 kg $CH_4 \cdot ha^{-1}$ BS versus 47.35 kg $CH_4 \cdot ha^{-1}$ in AS). With the straw removal, there is less organic matter decomposed into the soil and, consequently, methane emissions are reduced.

As mentioned in Section 2.8, a sensitivity analysis was performed on the economic allocation applied in the AS. The variation of rice and straw prices had a slight effect on the environmental results: When minimum prices were taken into account, an increase of 2.87% of the environmental burden associated to 1 FU was produced; instead, maximum prices determined a reduction of the impacts of 4.88%. Such results suggest that, notwithstanding price variation, the economic allocation could be a reliable approach when the environmental impacts need to be allocated between rice and straw.

4. Discussion

The environmental load of rice cultivation is mainly due to fertilization. In fact, the production of mineral fertilizers is an energy-intensive process that involves a high environmental impact and the application of fertilizers, both mineral and organic, involves emissions from the soil.

The results obtained regarding the impact of the mechanization of field operations suggest that, for this aspect, the diesel fuel consumption

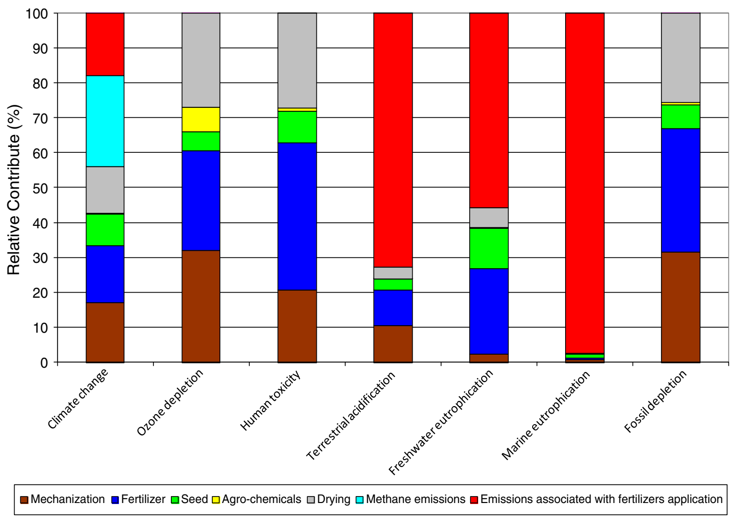


Fig. 4. Environmental impact for the alternative scenario (AS).

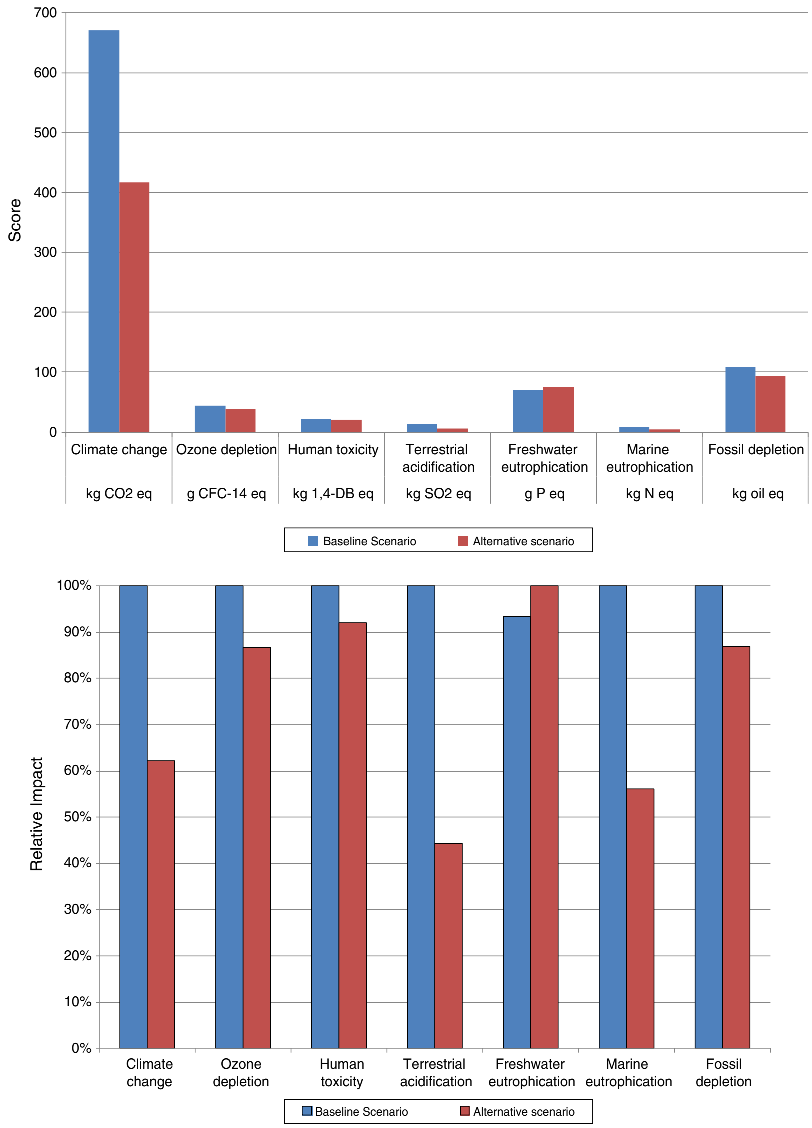


Fig. 5. Comparison between BS and AS; on the top absolute values; on the bottom, relative values.

plays a key role. In fact, the field operations with the higher environmental impacts are characterized by high fuel consumption. This result is in agreement with other studies focused on the assessment of the environmental performances of crop cultivation (Bacenetti et al., 2013; Gonzalez-Garcia et al., 2013; Bessou et al., 2013: Bacenetti et al., 2014; Gan et al., 2011; Kroebel et al., 2013).

Table 6
Rice yield for years 2009–2013.

Year	Yield[a]
2009	7.01
2010	6.36
2011	6.33
2012	6.78
2013	6.56

[a] Grain yield ($t \cdot ha^{-1}$) at 14% of moisture.

Concerning the possibility to up-scale the achieved results at other areas, it must be underlined that in the present study no chemical control against red rice has been considered. Moreover, the integrated production protocol adopted by Region Piedmont, with regard to rice cultivation, does not allow chemical application for pest control and limits also the use of fungicides and herbicides. For these reasons, pesticide applications have a lower impact on the environmental burdens than fertilizer application and fuel consumption. In areas with a strong red rice presence and/or years with climatic conditions favorable to the development of diseases (e.g. *Fusarium* spp.) and pests (*Lissorophtus oryzophilous*; *Hydrellia griseola*) the impact of pesticide application will be greater.

In our study the environmental load derived from the application of plant protection products is low if compared to other studies specifically focused on the evaluation of pesticide risk assessment in rice paddy. The extensive application of plant protection products (mainly herbicide and pesticides) in combination with wrong agricultural practices could result in environmental issues such as contamination of natural resources and risks for human health (Capri and Karpouzas, 2007; Beckie et al., 2014). The difference in the results could be due to the methods used for the assessment of pesticide application. Concerning the use of the LCA methodology to evaluate the impact from pesticide applications, some limits related with the methodology must be underlined to better understand why substances that are known to be connected to real environmental problems may receive low scores in the LCA environmental profile. LCA leaves risks and average human intake rates out of consideration. Environmental emissions are directly combined with environmental residence times to deliver time-integrated exposure totals. This makes LCA results independent of dilution, not only in the spatial sense, but also in terms of time. Environmental threshold values – that are so determinative for risk assessment – are not accounted for in LCA (Sleeswijk et al., 2000).

The emissions in LCA studies are assessed on the basis of a combination of potential harmfulness and environmental residence time but the preeminent role is played by bulk emissions. Environmental concentrations are not coupled to the total emissions that are responsible of the LCA environmental load.

Therefore, the prioritization of desired environmental measures should not be based on LCA results alone. LCA scores form an indication, but should be checked with measurements or by other evaluation methods (e.g. risk assessment) before conclusions on corresponding risk levels can be drawn. In this regard, specific guidelines for pesticide risk assessment in rice paddy have been recently developed taking into account the rice cultivation conditions (Capri and Karpouzas, 2007).

Regarding the identification of the environmental *hotspots*, our analysis shows similar results to other rice LCA studies carried out in temperate regions (Blengini and Busto, 2009; Drocourt et al., 2012; Fusi et al., 2013). In particular, Drocourt et al. (2012) evaluated the rice cultivation in Camargue (France) and highlighted that the fertilization is the most polluting process for all impacts, while methane emissions are the key aspect for CC. Also Hokazono and Hayashi (2012) and Hatcho et al. (2012), who evaluated rice cultivation in Japan, detected fuel consumption and methane emissions as key aspects for CC, while emissions associated with fertilizer application were the main contributor for acidification and eutrophication.

Concerning the Italian rice, Blengini and Busto (2009) carried out an LCA study of the whole productive process of rice in the Vercelli District and they identified the same hotspots: fertilizer application, methane emissions, and emissions associated with fertilizer applications. Nevertheless, the results are not comparable because the functional unit is different as is the system boundary considered.

From the analysis carried out, it can be stated that the approach chosen to handle the straw could heavily affect the results. If straw is collected (to be sold) and not incorporated into the soil, the impact of the environmental categories decreases, except for freshwater eutrophication. Higher mineral fertilizer applications, however, are still needed to compensate for the higher uptake of N, P, and K. This is due to the allocation between rice paddy and straw and to a sensible reduction of methane emissions. However, the proposed alternative solution for straw management can be carried out only when there is the possibility to valorize economically this by-product. It cannot be feasible easily in areas characterized by reduced livestock activity. Furthermore, in the long term, the effects related to the straw removal on the soil carbon content should be carefully evaluated.

4.1. Time analysis

Over the years, as a consequence of climatic conditions, the yields of paddy rice have varied. In this section, the average productions of rice in the District of Vercelli have been taken into account over five years (2009–2013). The different yields for the five years taken into account are listed in Table 6. A comparison of the results obtained for the different years was carried out: The changes in the results were from 0.5% up to 6.7%, maximum. By setting the higher impacts obtained (2011) equal to 100%, all other impacts are equal to 90.3% in 2009, 99.5% in 2010, 93.4% in 2012, and 97.4% in 2013.

As expected, the variations of yields over the years can affect the results. Yield variations affect the environmental burdens because of differences in the mass of useful product (the rice paddy) and the amount of straw production and, consequently, nutrient removal from the soil.

5. Conclusions

The evaluation of the environmental performance of agricultural activities is more and more important because agriculture is responsible for remarkable environmental impacts. Regarding the crop cultivations, this evaluation can help to detect the most environmentally friendly solutions.

For all the cereal crops, their cultivation can be carried out following different cultivation practices. The agronomical techniques and the farming managements exert a considerable influence on the environmental burdens. Also for the rice, a wide set of cultivation techniques can be carried out; the main differences are water management, tillage operations, fertilization, and straw management. These differences can be identified among different districts, while within the same district a standard cultivation technique can be identified.

In this paper, the standard agricultural practice (SCP) for rice cultivation in the District of Vercelli (Northern Italy) was detected and its environmental impact was analyzed using the LCA method. The analysis showed that the environmental impact was mainly due to the field emissions, the fuel consumption needed for mechanization of field operations, and the drying of the paddy rice. The comparison between two scenarios regarding different straw managements highlighted that the straw collection improves the environmental performance of rice production, except that for freshwater eutrophication.

To improve the environmental performances of the rice SCP, solutions that save fossil fuels, reduce the emissions from fertilizer use (leaching, volatilization), and lower methane emissions should be implemented.

The results obtained could be used for comparisons with, on the one hand, rice cultivations in other districts and, on the other hand, different cultivation practices carried out in the studied area.

In conclusion, the LCA method is a reliable tool for the assessment of the environmental load of agricultural processes; nonetheless it should be noted that for many impact categories (such as CC, OD, TA, FE and ME) the impacts are totally dominated by the bulk emissions of few substances or substance groups into the air (CO_2, CH_4, SO_2, NO_x, NH_3, PM_{10}, NMVOC) and emissions of N- and P-based compounds into fresh water. While, for the toxicity-related impact categories (HT in this study), the availability of information regarding the derived emissions from pesticides, metal compounds and some specific inorganics is still very limited, leading to large uncertainty in the corresponding results. To this regard, the importance of efficient measures must be stressed to combat bulk emissions and to promote the registration of potentially toxic emissions on a more comprehensive scale. For example besides the application of LCA, for a more comprehensive evaluation of negative effects on the environment due to rice cultivation risk assessment should be evaluated. In this way negative effect underestimated by LCA application (mainly based on a massive assessment) could be better highlighted.

Acknowledgments

The authors thank Regione Lombardia which financed a Postdoctoral Research Fellowship ("Progetto Dote Ricerca" financed by FSE — Regione Lombardia).

References

Althaus HJ, Chudacoff M, Hischier R, Jungbluth N, Osses M, Primas A. Life cycle inventories of chemicals. Ecoinvent report no. 8, v2.0 EMPA. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories; 2007.

Andersson K. LCA of food products and production systems. Int J Life Cycle Assess 2000; 5(4):239–48.

Bacenetti J, Negri M, Fiala M, González-García S. Anaerobic digestion of different feedstock: impact on energetic and environmental balances of biogas process. Sci Total Environ 2013;463–464:541–51.

Bacenetti J, Fusi A, Negri M, Guidetti R, Fiala M. Environmental assessment of two different crop systems in terms of biomethane potential production. Sci Total Environ 2014; 466–467:1066–77.

Baldoni R, Giardini L. Coltivazioni Erbacee. Cereali e proteoleaginose. Patron Editore; 2000. p. 1–409.

Beckie HJ, Sikkema PH, Soltani N, Blackshaw RE, Johnson EN. Environmental impact of glyphosate-resistant weeds in Canada. Weed Sci. 2014;62(2):385–92.

Bessou C, Basset-Mens C, Tran T, Benoist A. LCA applied to perennial cropping systems: a review focused on the farm stage. Int J Life Cycle Assess 2013;18(2):340–61.

Blengini GA, Busto M. The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). J Environ Manage 2009;90:1512–22.

Boschetti M, Bocchi S, Stroppiana D, Brivio AP. Estimation of parameters describing morpho-physiological features of Mediterranean rice varieties for modelling purposes Italian. J. Agrometeorol 2006;3:40–9.

Brentrup F, Küsters J, Lammela J, Barraclough P, Kuhlmann H. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur J Agron 2004;20:265–79.

Buratti C, Fantozzi F. Life cycle assessment of biomass production: development of a methodology to improve the environmental indicators and testing with fiber sorghum energy crop. Biomass Bioenergy 2010;34(10):1513–22.

Capri E, Karpouzas D. Pesticide risk assessment in rice paddies: theory and practice. Elsevier; 2007. p. 266.

CEC (Commission of the European Communities). Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Integrated product policy. Building on environmental life-cycle thinking. COM(2003)302 final. Brussels, Belgium: CEC; 2003.

Cellura M, Longo S, Mistretta M. Life Cycle Assessment (LCA) of protected crops: an Italian case study. J Cleaner Prod 2012;28:56–62.

Drocourt A, Mervant Y, Milhau F, Chinal M, Hélias A. Environmental assessment of rice production in Camargue, France. Paper presented at the 8th Conference on LCA in the Agri-Food Sector, Saint-Malo, France; 2012. p. 824–5.

Enterisi. Riso – Evoluzione di mercato e sue prospettive. Ente Nazionale Risi, Roma, 30 Novembre 2012; 2012.

Enterisi. Riso – Evoluzione di mercato e sue prospettive. Ente Nazionale Risi, Roma, 30 Dicembre 2013; 2013.

Eshun JF, Apori SO, Wereko E. Greenhouse gaseous emission and energy analysis in rice production systems in Ghana Afr. Crop Sci. J 2013;21(2):119–25.

Fallahpour F, Aminghafouri A, Ghalegolab Behbahani A, Bannayan M. The environmental impact assessment of wheat and barley production by using life cycle assessment (LCA) methodology. Environ Dev Sustain 2012;14:979–92.

FAO. FAO statistical yearbook 2013 world food and agriculture. Rome: Food and Agriculture Organization of the United Nations 978-92-5-107396-4; 2013a.

FAO. Rice market monitor. Trade and Markets Division, Food and Agriculture Organization of the United Nations, Volume XVI — Issue No. 3. ; 2013b. [available at http://www.fao.org/docrep/018/ar423e/ar423e.pdf].

Fiala M, Bacenetti J. Model for the economic, energetic and environmental evaluation in biomass productions. J Agric Eng 2012;42:26–35.

Frischknecht R, Jungbluth N, Althaus H-J, Bauer C, Doka G, Dones R, et al. Implementation of life cycle impact assessment methods. Ecoinvent report no. 3, v2.0. Dübendorf: Swiss Centre for Life Cycle Inventories; 2007.

Fusi A, González-García S, Bacenetti J, Feijoo G, Fiala M, Guidetti R, et al. Assessing environmental impacts of rice production in Italy. SETAC, 19th Case Study Symposium LCA in Market, Research and Policy: Harmonisation Beyond Standardisation. Rome, November 2013; 2013. p. 101–3. [2310–3191].

Gan Y, Liang C, Wang X, McConkey B. Lowering carbon footprint of durum wheat by diversifying cropping systems. Field Crop Res 2011;122:199–206.

Glithero NJ, Ramsden SJ, Wilson P. Farm systems assessment of bioenergy feedstock production: integrating bio-economic models and life cycle analysis approaches. Agr Syst 2011;109:53–64.

Goedkopp M, Heijungs R, Huijbregts M. ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Amersfoort, Netherlands: PRé Consultants, University of Leiden, Radboud University Nijmegen, RIVM, Bilthoven; 2009.

Goglio P, Bonari E, Mazzoncini M. LCA of cropping systems with different external input levels for energetic purposes. Biomass Bioenergy 2012;42:33–42.

González-García S, Bacenetti J, Murphy RJ, Fiala M. Present and future environmental impact of poplar cultivation in Po Valley (Italy) under different crop management systems. J Clean Prod 2012;26:56–66.

González-García S, Bacenetti J, Negri M, Fiala M, Arroja L. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J Clean Prod 2013;43:71–83.

Grab R, Heuser F, Stulpnagel R, Piepho H, Wachendorf M. Energy crops production in a double-cropping systems: results from an experiment at seven sites. Eur J Agron 2013;51:120–9.

Harada H, Kobayashi H, Shindo H. Reduction in greenhouse gas emissions by no-tilling rice cultivation in Hachirogata polder, northern Japan: life-cycle inventory analysis. Soil Sci. Plant Nutr. 2007;53:668–77.

Hatcho N, Matsuno Y, Kochi K, Nishishita K. Assessment of environment-friendly rice farming through life cycle assessment (LCA) CMU. J Nat Sci 2012;11(1). [Special Issue on Agr. Nat. Res.].

Hokazono S, Hayashi K. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. J Clean Prod 2012;28:101–12.

IPCC. IPCC guidelines for national greenhouse gas inventories. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. Prepared by the National Greenhouse Gas Inventories Programme. Japan: IGES; 2006.

ISO 14044. Environmental management – Life Cycle Assessment – requirements and guidelines. International Organization for Standardization; 2006.

Jungbluth N, Chudacoff M, Dauriat A, Dinkel F, Doka G, Faist Emmenegger M, et al. Life cycle inventories of bioenergy. Ecoinvent report no. 17. Dübendorf: Swiss Centre for Life Cycle Inventories; 2007.

Kanta Gaihre Y, Wassmann R, Tirol-Padre A, Villegas-Pangga G, Aquino E, Kimball BA. Seasonal assessment of greenhouse gas emissions from irrigated lowland rice fields under infrared warming. Agr Ecosyst Environ 2014;184:88–100.

Kröbel R, Janzen HH, Beauchemin KA, Bonesmo H, Little SM, McAllister TA. A proposed approach to estimate and reduce the environmental impact from whole farms. Acta Agric Scand Sect A Anim 2013;62(4):225–32.

Leip A. Contribution of rice production to of greenhouse gas emission in Europe. In: Bocchi S, Ferrero A, Porro A, editors. Proceedings of the Fourth Temperate Rice Conference. 25–28 June 2007, Novara, Italy; 2007.

Leip A, Bocchi S. Measurement of greenhouse gas emission from rice cultivation in Italy. In: Bocchi S, Ferrero A, Porro A, editors. Proceedings of the Fourth Temperate Rice Conference. 25–28 June 2007, Novara, Italy; 2007.

Margni M, Rossier D, Crettaz P, Jolliet O. Life cycle impact assessment of pesticides on human health and ecosystems. Agr Ecosyst Environ 2002;93(1–3):379–92.

Murphy C, Kendall A. Life cycle inventory development for corn and stover production systems under different allocation. Biomass Bioenergy 2013;58:67–75.

Nalley L, Popp M, Fortin C. The impact of reducing greenhouse gas emissions in crop agriculture: a spatial and production-level analysis. Agric Res Econ Rev 2011;40(1): 63–80.

Negri M, Bacenetti J, Brambilla M, Manfredini A, Cantore C, Bocchi S. Biomethane production from different crop systems of cereals in Northern Italy. Biomass Bioenergy 2014;63: 321–9.

Nemecek T, Käggi T. Life cycle inventories of agricultural production systems. Final report ecoinvent v2.0 No. 15a. Agroscope FAL Reckenholz and FAT Taenikon. Zurich and Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories; 2007.

Nemecek T, Huguenin-Elie O, Dubois D, Gaillard G, Schaller B, Chervet A. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agr Syst 2011;104(3): 233–45.

Neue HU, Wassmann R, Lantin RS, Alberto CR, Aduna JB, Javellana AM. Factors affecting methane emission from rice fields. Atmos Environ 1996;30(10–11):1751–4.

Roer AG, Korsaeth A, Henriksen TM, Michelsen O, Strømman AH. The influence of system boundaries on life cycle assessment of grain production in central southeast Norway. Agr Syst 2012;111:75–84.

Romani M, Beltarre G. Comparison of different sowing types and nitrogen fertilization in organic rice in Italy. In: Bocchi S, Ferrero A, Porro A, editors. Proceedings of the Fourth Temperate Rice Conference. SIRFI; 2007. p. 25–8. [June 2007, Novara, Italy.].

Roy P, Shimizu N, Okadome H, Shiina T, Kimura T. Life cycle of rice: challenges and choices for Bangladesh. J Food Eng 2007;79:1250–5.

Roy P, Ijiri T, Nei D, Orikasa T, Okadome H, Nakamura N, et al. Life cycle inventory (LCI) of different forms of rice consumed in households in Japan. J Food Eng 2009a;91:49–55.

Roy P, Nei D, Orikasa T, Xu Q, Okadome H, Nakamura N, et al. A review of life cycle assessment (LCA) on some food products. J Food Eng 2009b;90:1–10.

Sleeswijk AW, van Oers L, Guinée G, Struijs J, Huijbregts M. Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000. Sci Total Environ 2000;390:227–40.

Smil V. Phosphorus in the environment: natural flows and human interferences. Annu Rev Energy Environ 2000;25:53–88.

Spielmann M, Bauer C, Dones R. Transport services. Ecoinvent report no. 14. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories; 2007.

Wang B, Neue HU, Samonte HP. Role of rice in mediating methane emission. Plant Soil 1997;189:107–15.

Weiss F, Leip A. Greenhouse gas emissions from the EU livestock sector: a life cycle assessment carried out with the CAPRI model. Agr Ecosyst Environ 2012;149: 124–34.

Yoo SH, Choi JY, Lee SH, Kim T. Estimating water footprint of paddy rice in Korea. Paddy Water Environ 2013;12:43–54.

Zamagni A, Masoni P, Buttol P, Raggi A, Buonamici R. Finding life cycle assessment research direction with the aid of meta-analysis. J Ind Ecol 2012;16:39–52.

THE ENVIRONMENTAL IMPACT OF BAGGED SALAD AND THE BENEFITS ARISING FROM WATER REUSE: A CASE STUDY

A. Fusi[a*], V. Castellani[b], J. Bacenetti[a], G. Cocetta[a], M. Fiala[a], R. Guidetti[a]

[a] Department of Agricultural and Environmental Sciences - Production, Landscape, Agroenergy, Università degli Studi di Milano, Via G. Celoria 2, 20133 Milano, Italy

[b] Department of Earth and Environmental Sciences, University of Milano-Bicocca, Piazza della Scienza 1, 20126 Milano, Italy

* Email: alessandra.f.m.fusi@gmail.com; Tel. +39 02 50316873; Fax. +39 02 50316845

Abstract

Purpose Fresh-cut processed vegetables are defined as those subjected to some processing techniques of lesser magnitude than canning or freezing, but which, nevertheless, added value to the product.

The fresh-cut market represents about 18% of the entire economic value of the fruit and vegetable market in Italy, and 2% of the total food market. Over the past two decades, worldwide awareness regarding environmental issues has consistently increased: environmental aspect is now one of the variables taken into consideration by consumers during the purchasing process. The aim of this study is to evaluate the environmental burdens of one bag of fresh-cut salad in order to identify the most critical production phases and suggest possible improving solutions.

Methods Such evaluation is been base on is based on the ISO standards for life

cycle assessment. The selected functional unit is an "average bag" containing 130 grams of fresh cut lamb's lettuce. Data concerning field operations, processing phases and transportation to the logistic platforms were obtained directly from the producer, while background and foreground data come from Ecoinvent.

Results and discussion Results show that, even though the agricultural activities can be environmentally relevant due to the high use of diesel fuel for agricultural machinery and the use of chemicals as fertilisers and pesticides, the environmental performance of the product analysed is mainly affected by the washing and packaging phases. This is due to the higher consumption of energy and the use of water and related production of wastewater, which need to be treated. Considering that the water consumption for cleaning is responsible for a large environmental impact, the possibility to install a filtration plant for the recovery of 40% of the washing solution has been evaluated. The reduction of the environmental impact stemming from the introduction of the water filtering system is considerably relevant for some categories.

Conclusions This case study highlights the environmental burdens associated with the current food production chains, which have become more complex during the last decades. The need to ensure food safety and conservation for longer periods and longer transport distances, and the increasing demand of ready-to-eat food imply the need for longer and more complex production chains. The agricultural phase is only the first step, instead of being a self-standing chain as it was in the past.

Keywords: fresh-cut salad, Life Cycle Assessment, water reuse.

1. Introduction

The term 'baby leaf' refers to vegetables harvested during the early stages of development, when they have reached a height of approximately 12 cm (Castoldi et al., 2011). The cultivation of baby leaf takes place during the whole year and is carried out in green houses. During summer, the production cycle of baby leaves can be as short as 20 days; however, during winter, the production cycle may exceed 40 days.

Fresh-cut, processed vegetables are defined as those subjected to some processing techniques of a lesser magnitude than canning or freezing, but which, nevertheless adds value to the product (Martín-Diana et al., 2006). In recent years, fresh-cut vegetables have become of great interest among consumers because they represent a very practical alternative to traditional vegetable crops and are recognised as healthy foods with a high adding-value (Torrellas et al., 2012). The sales of fresh-cut vegetables have grown rapidly in the last years as a result of changes in consumer attitudes (Rico et al., 2007). Nowadays, consumers are more aware of quality aspects of produce. Compared to traditional products, the fresh-cuts offer a wide number of advantages, such as freshness, safety, practicality and labelled information (expiration date, nutritional aspects and weight) (Ragaert et al., 2004). Moreover, these aspects lead to a higher price and so to economic advantages for those involved in the production chain. Therefore, fresh-cut, minimally processed, and ready-to-eat leafy vegetables have gained importance in Italy and Europe in recent years (Casati and Baldi, 2012). In Italy, the area of cultivation is approximately 6500 ha, and 70% of all fresh-cut cultivation is conducted in greenhouses by 450 farms (Castoldi et al., 2011).

The fresh-cut market represents about 18% of the whole economic value of the fruit and vegetable market in Italy, and the 2% of the total food market. In terms of mass, salads represent 70% of the fresh-cut market; among these, the baby leaves account for 40% while other adult plants cover 60% of the market (Casati and Baldi, 2011). As it is for any "ready-to-eat" product, the production process of fresh cut baby leaves entails several further stages after the agricultural phase. Following the harvest, the fresh-cut production consists of cutting, cleaning, disinfection and packaging. All these phases represent an additional source of environmental impacts (emissions and energy and water consumption) (Olmez and Kretzschmar, 2009) and represent a pivotal challenge for this sector.

There is increased awareness that the environmentally conscious consumer of the future would include ecological and ethical aspects in the purchasing decision-making process (Roy et al., 2008; Pluimers, 2011; Poritosh et al., 2009). Thus, it is essential to evaluate the environmental impact and the use of resources in food production and distribution systems.

LCA is a standardised process for evaluating the environmental burdens associated with a specific product, for estimating consumption of natural resources and emissions to environmental compartments and for identifying and implementing opportunities to attain environmental improvements. Although, in the beginning, it was developed for the evaluation of industrial processes, today the LCA is more applied also to agricultural activities. Over the years, LCA has been used for the evaluation of the environmental performances of annual and perennial crops (Blengini and Busto, 2009; Gonzalez-Garcia et al., 2013a; Bacenetti et al., 2014; Fusi et al., 2014), bio-energy chains (Dressler et al., 2012; De Vries et al., 2012; Bacenetti et al., 2013) and industrial food products (Antón et al., 2005; Bevilacqua et al., 2007; Hospido et al., 2009; Roy et al., 2009; Milà I Canals et al., 2010; Martinez-Blanco et al., 2011; Amienyo et al., 2013; Gonzalez-Garcia et al., 2013b). So far, few studies have been published on the evaluation of the environmental impact associated with the production of baby leaf vegetables, and some of those studies were mainly focused on greenhouse gas (GHG) emissions and fossil energy consumption (Castoldi et al., 2011).

The aim of this study is to evaluate, by the LCA methodology, the environmental burdens of the baby leaf production system carried out in greenhouses in Northern Italy, and to investigate the further phases of cleaning and packaging carried out to obtain a ready-to-eat product. As the main subject of our research, we chose lamb's lettuce (Valerianella locusta L.), as it is a widespread cultivation in Northern Italy, and because it is an important leafy vegetable which provides one of the highest amount of antioxidant compounds among leafy vegetables (Müller, 1997; Ferrante et al., 2009). From a nutritional and technological point of view, in fact, thanks to the high content of antioxidant compounds, lamb's lettuce shows a good attitude toward fresh-cut processing together with a good storability. Moreover, compared to other species, in the case of lamb's lettuce, cutting operations are reduced, as the whole seedling is commonly used, allowing for the maintenance of a higher quality over the whole production chain.

The hotspots of the system have been identified and discussed in detail. Considering that the water consumption for cleaning is responsible for a large environmental impact, the possibility to install a filtration plant for the recovery of 40% of the washing solution has been evaluated in order to assess the environmental benefits arising from water reuse.

2. Materials and methods

2.1 Goal and scope of the study

The aim of this study is twofold:

1. Evaluating the environmental performances of one bag of fresh-cut salad produced by an Italian company using the LCA methodology;
2. Analysing the environmental benefits deriving from the introduction of a water filtering system in the processing phase.

An LCA analysis was, therefore, performed on a bag of fresh-cut salad. In order to pursue the last goal, two scenarios were assessed:

1. No water reuse (NWR) scenario, in which the processing water was not subject to a filtering phase; this scenario represented the current condition of the processing plant;
2. Water reuse (WR) scenario, in which 40% of the processing water was reused in the plant after filtering. The percentage of water reuse, as well as the energy savings derived from the introduction of a water filtering system, were provided by the company involved in the analysis.

2.2 The production process

The production system takes place in the Lombardy region, the most populated region of Northern Italy.

The production pipeline can be divided into three subsystems:

1. Greenhouse production under plastic tunnels (not heated): this subsystem involves the agricultural practices carried out in the greenhouses. The greenhouses, located in Treviglio (District of Bergamo), are built with polyethylene (PE) (lifespan six years) and galvanised iron. Each greenhouse has a global area of 480 m^2, of which 327 m^2 is cultivated with lamb's lettuce. During the year, five to six cultivation cycles are carried out. The crop cycle is shorter in spring (about 25–30 days) than in winter and summer (about 40–60 days). The lamb's lettuce has an optimal growing temperature from 18–22°C. Therefore, in summer and winter, the growing cycles are longer. In summer, from June to September, for avoiding excessive solar radiation, the greenhouses are shaded with black nets that reduce up to 60–70% of the solar radiation. Before sowing, once a year, an organic fertilisation with cow and horse manure is carried out in order to reach 2–2.5% of organic matter in the soil, as well as a soil tillage performed by means of a rotary harrow. The sowing is carried out by a pneumatic sowing machine. Pest and disease management involve one intervention by spreader for the distribution of an herbicide and two fungicides. Irrigation is carried out by means of pumps, which involves electricity consumption. The plants are harvested by hand tools in order to cut the whole plants under the soil because plants are commercialised with part of the roots, and the average yield is 0.8 kg/m^2.
2. Transport: after harvest, the fresh salad is transported inside plastic crates to the processing centre by a lorry (mass > 16 t). The distance between the greenhouse and the processing centre is 30 km; about 2700 kg of product is transported for each trip while, at the return, the lorry is empty. Each lorry is equipped with a refrigeration unit in order to keep the product refrigerated (4–5°C).
3. Processing: in this subsystem, the product is cleaned, sanitised with a solution of sodium hypochlorite and then packaged in plastic bags. The cleaning involves water and disinfectant consumption; well water refrigerated to 10–12°C by means of a chiller and sodium hypochlorite is added (30% of the water has a concentration of sodium hypochlorite equal to 50 mg L^{-1} while the 70% of it is 2 mg L^{-1}). Although it would be technically feasible by means of a filtration plant, actually, there is not water reutilisation. With the packaging, the fresh salad is wrapped in plastic bags (polyethylene); this operation is carried out by a specific machine, which is able to weigh a specific amount of product (from 80 to 200 grams). Figure 1 represents the processing flow-chart of fresh cut salad. The dashed arrow indicates the recycling of washing water considered in the WR scenario.

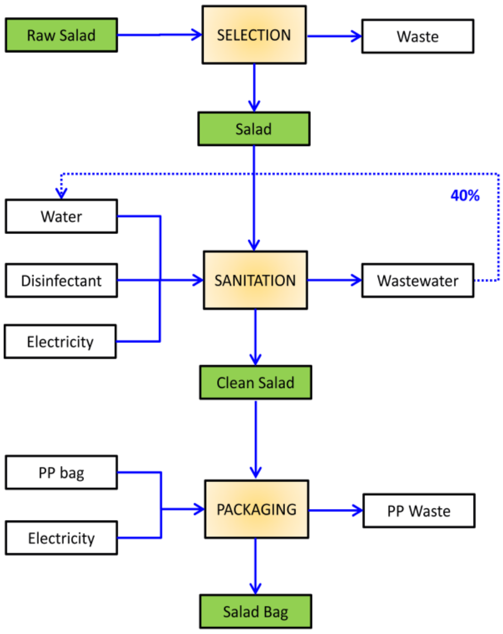


Fig 1 Processing flow-sheet of fresh cut lamb's lettuce

2.3 Functional unit

The selected functional unit (FU) was an "average bag" containing 130 grams of fresh cut lamb's lettuce *Locusta*. This quantity was obtained by dividing the total annual production of fresh cut lamb's lettuce by the total amount of bags produced in one year.

2.4 System boundaries

The following life-cycle steps were assessed (Fig. 2):

- Production of chemicals (fertilisers and pesticides) and seeds.
- Production of diesel fuel, electricity and water.
- Cultivation process, including the use of energy, water and materials during the various crop treatments and harvest.
- The delivery of salad to the processing plant.
- Salad selection, washing and packaging (primary packaging).The distribution of the packed baby leaf salad to the logistic platforms, (including secondary packaging).

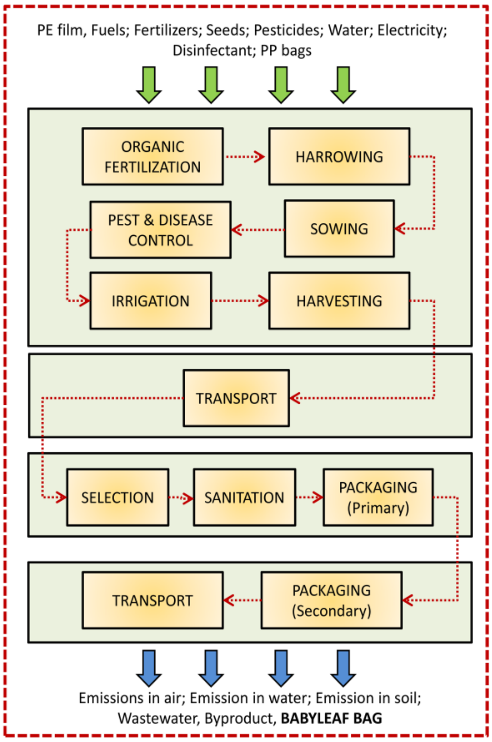


Fig 2 System boundaries

2.5 Inventory

Data concerning field operations, processing phases and transportation to the logistic platforms were obtained directly from the producer. Background and foreground data come from Ecoinvent v. 2.2 (Frischknecht et al., 2007), ELCD (European reference Life Cycle Database, Joint Research Centre), Industry database version 2.0 and DETEC.
Data refer to the year 2012, and are representative: the production is, in fact, constant over time.

2.5.1 LCI of agricultural phase

LCI (Life Cycle Inventory) data for the agricultural phase include fuel consumption for machinery use, emissions from fertilisers and pesticides and construction of greenhouses. Nitrogen emissions to air and water (dinitrogen monoxide (direct and indirect emissions) and ammonia) from the fertilisation phase were calculated according to the 2006 IPCC emission factors. Pesticide emissions depend on several factors, such as the way in which a pesticide is applied and the meteorological conditions during application (EMEP/EEA, 2013). Since detailed data about these factors were not available for the case study, pesticide emissions into air, water and soil were estimated in accordance with Margni et al. (2002) and Audsley et al. (1997). According to these studies, the fraction of active ingredients entering the soil is assumed to be 85% of the total applied quantity; 5% remains on the plant and 10% is emitted into the air. The run-off of the active ingredient from the soil into the water is assumed to be a maximum of 10% of the applied dose.
Regarding fuel use, the amount of diesel used by each machine for field operations was directly provided by the producer; the emissions generated (hereafter referred to as "diesel emissions) were instead estimated using data from the Swiss Federal Office for the Environment (Federal Department of the Environment, Transport, Energy and Communications, or DETEC).
Consistent with other studies (Cellura et al., 2012a; Benedetto, 2013; Fusi et al., 2014), CO_2 absorbed by the plants during their vegetative cycle were not taken into account. Regarding the greenhouse used for the cultivation phase, only the plastic film (LDPE) of the structure was considered. The reason behind this choice was the lifespan of the film (three years); it was smaller with respect to the metal structure and foundations (10 and 30 years, respectively, according to Cellura et al. (2012a)). This choice was made in accordance with the PCR (Product Category Rules) on

vegetables (CPC 012, v. 1.01, 2013), which establishes that "the manufacturing of production equipment with an expected lifetime over three years, buildings and other capital goods shall not be included."
Table 1 lists the agricultural operations carried out in the cultivation phase, the agricultural machineries used and their characteristics, as well as the yield of lamb's lettuce provided by the producer.
Table 2 reports a data inventory for the agricultural phase.

Table 1. Agricultural operations and agricultural machineries characteristics.

Operation	**Machinery**	**Fuel type or Energy**	**Consumption**	**Tractor power**	**Year of production (tractor)**
Pre-sowing fertilisation	Fertiliser spreader	Diesel fuel	5.72 kg	37 kW	2012
Soil tillage	Hoeing machine	Diesel fuel		95 kW	2009
Sowing	Manual	-	-	-	-
Plant protection	Sprayer	Diesel fuel	3.26 kg	37 kW	2012
Irrigation	Pump	Electricity	50 kWh	-	

Table 2. Data inventory for the agricultural phase (data related to FU).

		Description	**Units**	**Value**
Inputs	Seeds	-	mg	370.87
	Cow and Hourse Manure	Organic fertiliser	g	40.46
	Fosetyl-Al	Pesticide	mg	24.91
	[Thio]carbamate-compounds[a]	Pesticide	mg	42.59
	[Sulfonyl]urea-compounds[b]	Pesticide	mg	50.57
	Diesel fuel	Fuel	g	5.25
	Water	-	dm^3	24.16
	Electricity for irrigation	-	kWh	0.0281
	LDPE film (greenhouse)	-	mg	41.36
	Land	-	m^2	0.27
Outputs	Salad (*Valerianella locusta*)	-	g	147
Emissions to air	Carbon dioxide	From diesel fuel	g	19.71
	Carbon monoxide	From diesel fuel	mg	101.43
	Nitrogen oxides	From diesel fuel	mg	174.14
	Particulate	From diesel fuel	mg	15.12
	Hydrocarbons	From diesel fuel	mg	19.89
	Dinitrogen monoxide	From diesel fuel	mg	11.32
	Ammonia	From fertilizer	mg	122.82
	Benfluralin	From pesticide	mg	5.06
	Fosetyl-Al	From pesticide	mg	2.49
	Propamocarb	From pesticide	mg	4.26
Emissions to water	Benfluralin	From pesticide	mg	5.06
	Fosetyl-Al	From pesticide	mg	2.49
	Propamocarb	From pesticide	mg	4.26
Emissions to soil	Benfluralin	From pesticide	mg	37.93
	Fosetyl-Al	From pesticide	mg	18.78
	Propamocarb	From pesticide	mg	31.94

[a]Propamocarb
[b]Benfluralin

2.5.2 LCI of processing phase

Table 3 reports a data inventory for the NWR scenario. In the WR scenario, besides the reduction of water use, a lower amount of electricity is needed because of the reduced use of water pumps to extract water from the wells. The reduction of electricity consumption, as well as the percentage of the water recycled for the WR scenario, were provided by the company based on their estimates (Table 4). In this analysis, filters used in order to ensure the proper quality of recycled water were not considered due to the lack of information.

Table 3. Data inventory for the processing phase (data refer to FU).

Inputs	**Units**	**Value**
Salad	g	147
Electricity[a]	kWh	0,109
Water	dm^3	5.02
Sodium hypochlorite	mg	76
PP film	g	3.94
Outputs		
Salad bag (130g)	p	1
Salad scraps	g	17
PP film waste[b]	g	0.187
Wastewater	kg	5.02

[a] Electricity used both for the washing and packaging phases as well as for the refrigerated storage at the plant facility: a partition of consumptions was not possible since aggregated data were provided.
[b] 5% of the PP film used for packaging is discarded (the fate of this waste was drawn in accordance with CARPI, Consorzio Autonomo Riciclo Plastica Italia)

Table 4. Water and electricity input in the WR scenario (data refer to FU).

Input	**WR scenario**	**Reduction (%) with respect to NWR**
Water	3,012 kg	40
Electricity	0,104 kWh	4.7

2.5.3 LCI of transportation phase

The refrigerated transport from the processing plant to the logistic platforms was considered, as reported in Table 5. The life cycle inventory data for transport have been sourced from Ecoinvent (v. 2.2, Frischknecht et al., 2007), but have been modified to include the additional amount of fuel (and the emissions) used by the refrigeration unit. The additional amount of fuel needed by the refrigeration unit (+30% with respect to ambient transport) was estimated based on the information from DEFRA (2008).
In order to deliver the fresh cut salad to the logistic platforms, bags are packaged into cardboard or polypropylene (PP) boxes. In Table 6, the main characteristics of the secondary packaging are presented (information was provided by the manufacturer).

Table 5. Transport details.

Logistic platforms	**Distance (km)**	**Bags delivered (% of the total)**	**Secondary packaging**
1	50	51	PP box
2	75	18	PP box
3	60	9	Cardboard box
4	595	7	Cardboard box
5	135	5	PP box
6	200	5	PP box
7	95	4	PP box

Table 6. Secondary packaging data.

Secondary packaging	**Mass (kg)**	**Average capacity (number of salad bags)**	**Lifespan (years)**	**Usage cycles per year**
Cardboard box	0.5	7.3	disposable	-
PP box	1	10	7	10.24

2.5.4 Allocation

The processing phase originates, besides fresh cut salad's bags, with scraps of lamb's lettuce, which are given for free to a buffalo breeding camp. Due to its fate, this waste was considered a feed substitute for livestock (grass from a meadow). The system expansion approach was used for allocating environmental impacts of fresh cut salad and scraps: the use of lamb's lettuce scraps instead of grass as feed for livestock allows for avoiding of the production of grass cultivated for feed purposes (this substitution was made based on dry matter content). To gauge the influence of this allocation approach, a sensitivity analysis was carried out, as discussed below.

2.6 Impact assessment

SimaPro (version 7.3.3) was used to model the life cycle of fresh cut lamb's lettuce, using midpoint indicators. Consistent with other studies carried out on greenhouse crops, the following impact categories were selected to evaluate the environmental impact of the product under study: Climate Change (CC) or Global Warming Potential, Fossil Depletion (FD), Terrestrial Acidification (TA), Freshwater and Marine Eutrophication (FE and ME), Human Toxicity (HT), Terrestrial, Marine and Freshwater Ecotoxicity (TE, MEc and FEc), Ozone Depletion (OD), Photochemical Oxidant Formation (POF) and Water Depletion (WD) due to the intensive use of water during both the agricultural and processing phases. LCIA (Life Cycle Impact Assessment) was carried out using the ReCiPe method (Goedkoop et al., 2009).

2.6.1 Sensitivity analysis

In order to test the robustness of the results and to investigate the influence of the choices made in the modelling phase, a sensitivity analysis was carried out on the system under study. A set of parameters was changed, and its influence on the results was evaluated. The most uncertain parameters were taken into account to run the sensitivity analysis. Due to their relevance on the environmental impact, the parameters associated with the emission of nitrogen compounds due to fertiliser use in the agricultural phase were considered. The emission factors for fertiliser emissions were modified within the range defined by IPCC (minimum and maximum value). The analysis was carried out considering, firstly, all minimum values of emission factors and, secondly, all maximum values (Table 7). The assumptions made on pesticide emissions underwent a sensitivity test as well; keeping constant the percentage of pesticides absorbed by the plant (5%), the fraction of active ingredients released in the soil, air and water was changed accordingly with the data provided by EMEP/EEA, 2013. The percentage of pesticides emitted into the air was, therefore, increased up to 25%, while the run-off of the active ingredient from the soil into the water was decreased to 5% of the applied dose. The emissions into the soil could, therefore, just account for 65% of the active ingredient. Moreover, in order to evaluate the robustness of the model, a scenario without allocation was evaluated. The sensitivity analysis was performed on the NWR scenario.

Table 7. Parameters and respective changes considered in the sensitivity analysis for fertilisers.

		Default value	**Range**	
			Min	**Max**
		kg N_2O-N/kg N		
Fertilizer use	N_2O emission factor from all N inputs (direct emissions)	0.01	0.003	0.03
	N_2O emission factor from N volatilization and re-deposition	0.01	0.002	0.05
		Share of N which is transferred		
	Volatilization for synthetic fertilizer	0.10	0.03	0.30
	Volatilization for organic fertilizer	0.20	0.05	0.50

3. Results and discussion

Figure 3 presents the results of one fresh-cut salad bag along all the life cycle stages taken into account in this analysis: the production phase, which includes both the agricultural and the processing phase, and the distribution phase, separated into secondary packaging (plastic and cardboard boxes) and the transportation itself. The environmental burden of transportation appeared to be negligible, as it contributed 1.9% maximum (OD) to the overall environmental impact. The majority of the product (78%) is, in fact, delivered within a distance of 75 km from the production site. The secondary packaging, on the other hand, had a slightly higher impact, especially when cardboard boxes were used. Cardboard contributes 15% to the FEc category and over 10% to HT, MEc and FE. PP boxes contributed instead a maximum of 8.6% on OD. The burden associated with PP boxes was lower with respect to disposable cardboard boxes by virtue of their reusability. As expected, the major contribution to the environmental impact of one fresh-cut salad bag, for all impact categories, was attributable to the production phase, which is examined below.

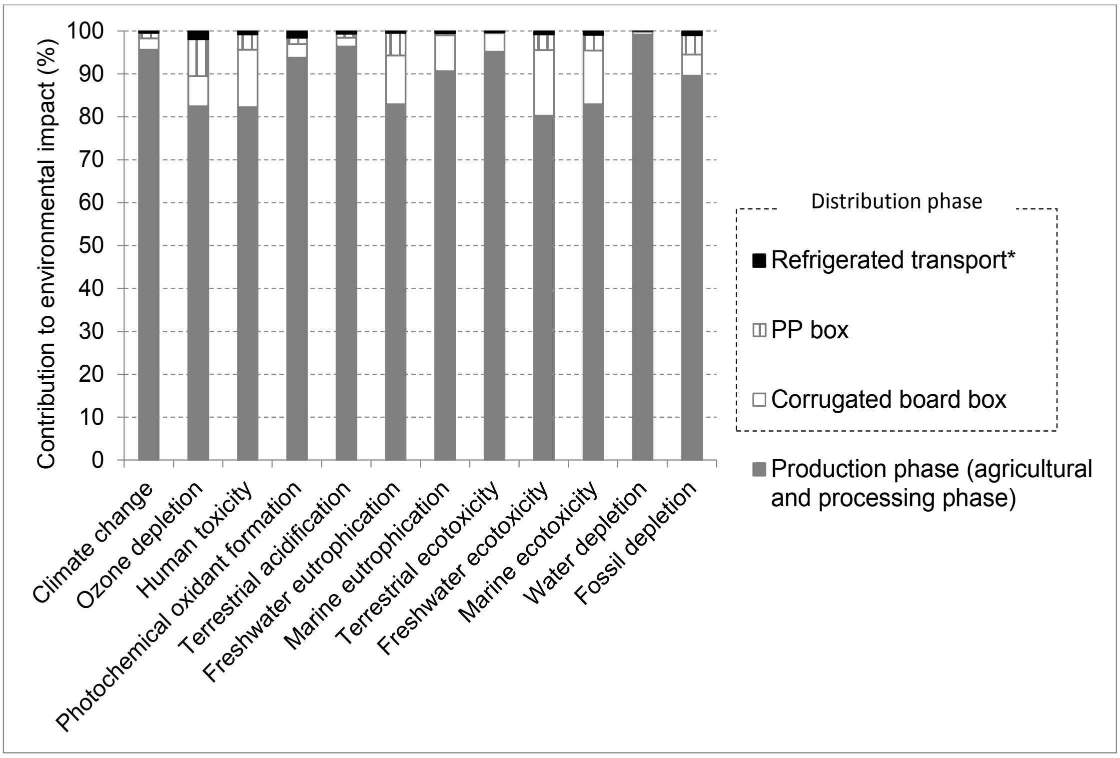


Fig 3 Contribution of each phase (production and distribution) to produce one FU
[*Transport includes: operation of vehicle; production, maintenance and disposal of vehicles; construction and maintenance and disposal of road; diesel fuel; emissions associated with transport.]

Table 8 reports the total and relative impact values per FU linked to the production phase of fresh-cut salad, namely the agricultural and the processing phase (washing and packaging). The processing phase generates greater impacts on 10 of 12 impact categories considered. The only impact categories for which the contribution from the agricultural stage was preponderant are TE and WD, which represent the most critical issues for agricultural activities.

The greater contribution to environmental impacts by the processing phase compared to the agricultural one is not surprising in the case of "ready-to-eat" products. Garnett (2006) listed ready-prepared, chilled fruits and vegetables (such as ready chopped fruit salads and bagged salads) among the highest GHG (Green House Gas) intensive vegetable products. The reason for such a result lies in the increased demands on the cold chain, in the additional washing and processing energy use and in the energy requirement for modified atmosphere packaging (if present). Even though there are no studies available on fresh-cut vegetables, the literature can provide some case of LCA studies conducted on ready-to-eat meals. The production process of meat or pasta ready meals is certainly more complex with respect to the ones of fresh-cut salad. Nevertheless, the analysis of these products could give an idea of how much the impact is of the processing phase for complex chains. The study carried out by Davis and Sonneson (2008) showed that, for a semi-prepared meal, post-farm activities account for 75% of the total amount of energy used along the whole chain. Berlin and Sund (2010) performed an LCA on various ready meals, revealing that processing raw materials can have quite an important impact on the whole life cycle of the products under analysis, depending on which kind of food was analysed. The processing phase accounted for 14% and 10% of global warming potential (GWP) for pork and pasta ready meals, respectively. The GWP associated with the factory stage for lamb ready meal was, instead, practically negligible. In the study carried out by Andersson et al. (1998) on tomato ketchup, the processing and the packaging subsystems make large contributions to global warming, while, for energy use, the storage time in a refrigerator (household phase) was found to be a hotspot.

Table 8. Results of the production phase (expressed in absolute values and in percentage of contribution) from the characterisation step presented for each impact category.

Impact categories	**Units**	**Total production phase**	**Agricultural phase**		**Transport from the greenhouse to the processing plant**		**Processing phase**	
		Value	*Value*	*Relative contribution on production phase (%)*	*Value*	*Relative contribution on production phase (%)*	*Value*	*Relative contribution on production phase (%)*
Climate change	kg CO2 eq.	3.02E-01	5.75E-02	19.03%	5.87E-04	0.19%	2.44E-01	80.77%
Ozone depletion	kg CFC-11 eq.	1.00E-08	4.61E-09	46.04%	9.48E-11	0.95%	5.31E-09	53.01%
Human toxicity	kg 1,4-DB eq.	1.55E-02	4.76E-03	30.67%	7.45E-05	0.48%	1.07E-02	68.85%
Photochemical oxidant formation	kg NMVOC	8.27E-04	2.76E-04	33.30%	5.73E-06	0.69%	5.46E-04	66.01%
Terrestrial acidification	kg SO2 eq.	1.25E-03	6.20E-04	49.51%	3.36E-06	0.27%	6.29E-04	50.22%
Freshwater eutrophication	kg P eq.	1.92E-05	6.05E-06	31.52%	5.65E-08	0.29%	1.31E-05	68.18%
Marine eutrophication	kg N eq.	8.98E-05	2.83E-05	31.56%	2.03E-07	0.23%	6.12E-05	68.22%
Terrestrial ecotoxicity	kg 1,4-DB eq.	5.31E-05	4.88E-05	91.80%	9.00E-08	0.17%	4.26E-06	8.03%
Freshwater ecotoxicity	kg 1,4-DB eq.	3.28E-04	1.09E-04	33.15%	1.60E-06	0.49%	2.18E-04	66.36%
Marine ecotoxicity	kg 1,4-DB eq.	3.35E-04	1.02E-04	30.59%	1.85E-06	0.55%	2.30E-04	68.85%
Water depletion	m^3	2.58E-02	2.43E-02	94.11%	2.36E-06	0.01%	1.52E-03	5.88%
Fossil depletion	kg oil eq.	4.95E-02	1.31E-02	26.38%	2.22E-04	0.45%	3.62E-02	73.17%

Figure 4 shows the impacts (in terms of relative values) of material and energy flows associated with the cultivation phase. The critical factors are fertiliser emissions (decisive for TA and ME), diesel fuel emissions (important for CC and POF), diesel fuel production (decisive, above all, for FD and OD), electricity, pesticide emissions (over 90% of TE) and water for irrigation (99% of WD). Even though the emissions related to the application of the organic fertiliser (compost) are far more relevant, its production is not negligible for some impact categories (CC, TA and ME): this is due to the emissions (such as CH_4, CO_2, NH_3) generated during the composting process. As for pesticides, their production is not critical. Attention should be paid to using the correct dose of pesticide in order to avoid the dispersion of active ingredients in the air, soil and run-off into the water. Emissions stemming from pesticides and fertiliser application show important contributions in different impact categories: TA and ME for organic fertiliser, and TE for pesticide. Human toxicity is not directly influenced by the emissions of pesticides, as the larger contributor for this impact category is electricity consumption. As illustrated before, since TE impact is mainly due to pesticide emissions, a sensitivity analysis was run in order to verify the influence of the assumptions made on the final results.

It is interesting to put the results of the agricultural phase of lamb's lettuce of this study into perspective with respect to the production of other types of salads, such as lettuce and endive, whose cultivation is actually more widespread. There are data sources for the latter, and they are mainly available for carbon footprint (see Table 9).

Table 9. Carbon footprint values for different types of salad (only the agricultural phase is considered).

Product	**Carbon footprint (kg CO_2 eq/kg salad)**	**Source**	**Inclusion of greenhouse production in system boundaries**
Lettuce	0.192	Venkat, 2012	No
	0.219	Romero-Gomez et al. (2014)	Yes
	0.32	Maraseni et al., 2010	No
	0.32	Hall et al., 2014	Yes
Iceberg lettuce	0.35	Davis et al. (2011)	No
Endive	1.17 – 1.75	Vieux et al., 2012	No
Escarole	0.28	Romero-Gomez et al. (2014)	Yes
Lamb's lettuce	0.395	Current study	No

Figure 5 reports impacts (in terms of relative and absolute values, respectively) of all inputs and outputs related to the ready-to-eat salad chain (distribution excluded). Besides the impact associated with the raw material (salad), the two major contributors are electricity (including electricity needed for refrigerated storage at the plant facility) and the wastewater produced (the impact of wastewater includes its treatment needed to clean it up). Wastewater has, nevertheless, a negative contribution on the WD category (since the wastewater treatment allows the water consumed to be available again), while the avoided product grass generates a positive effect of 5% maximum on the FEc. The sodium hypochlorite used as sanitiser in the processing phase appears not to contribute to the environmental impact of the salad bag as well as the PP film used for packaging, which has a slight influence on the overall results. The processing phase is dominated by the washing and packaging operations (due to the electricity and water consumption). Therefore, in spite of the growing concern about packaging material as one of the potential hot-spots in the food chain (e.g., Marsh and Bugusu, 2007; Williams and Wikström, 2011), packaging material itself is not so relevant, from an environmental perspective, in the production chain considered: this is because the phases needed to obtain a ready-to-eat product generate a higher environmental impact.

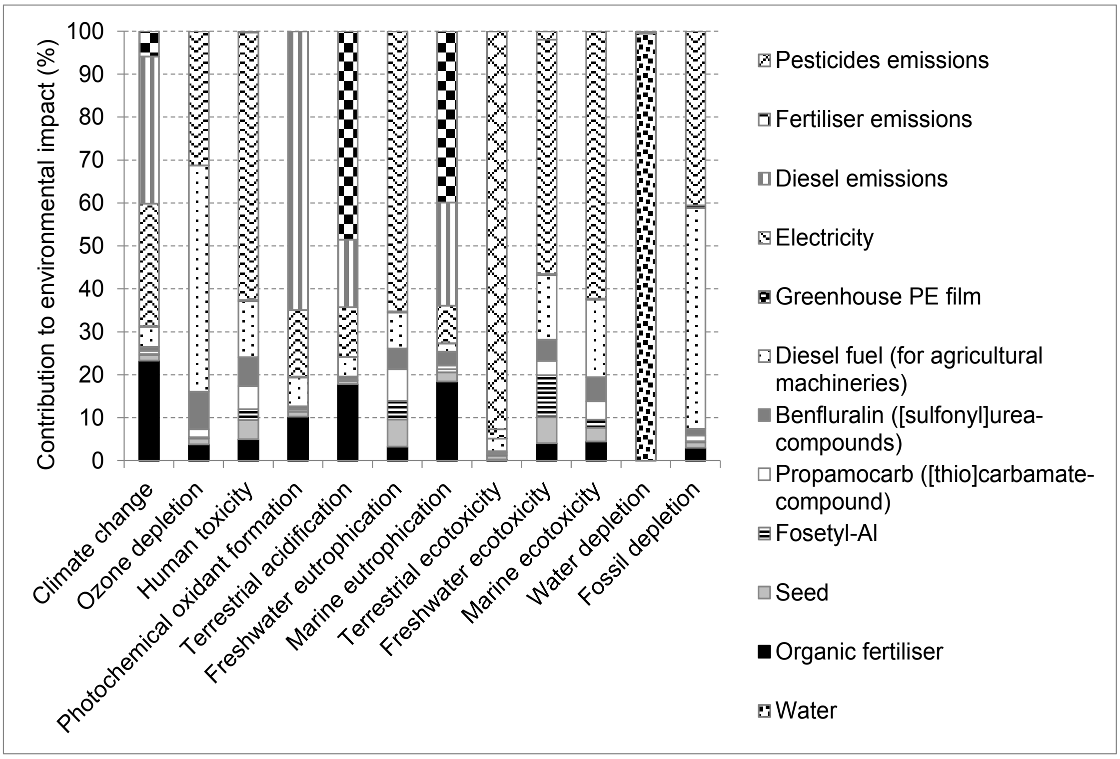


Fig 4 Contribution of each input and output in the agricultural phase

11

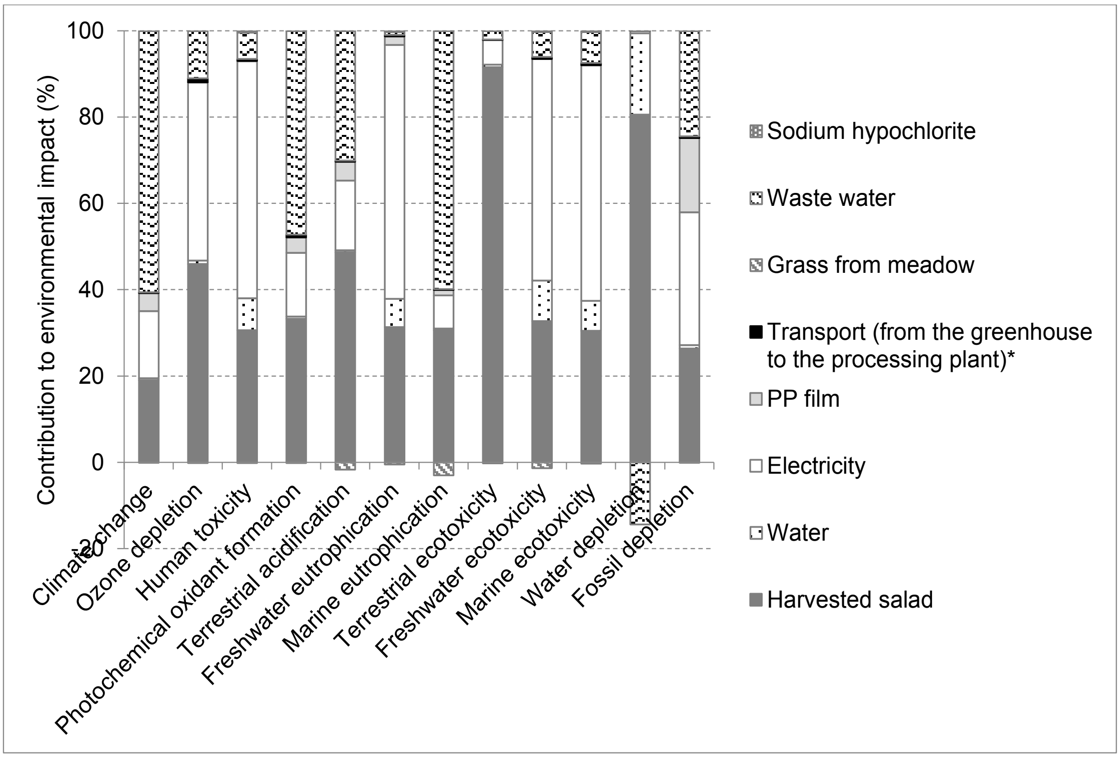


Fig 5 Contribution of each input and output in the production of 1 bag of fresh-cut salad
[*Transport includes: operation of vehicle; production, maintenance and disposal of vehicles; construction and maintenance and disposal of road; diesel fuel; emissions associated with transport.]

In order to reduce the impact from wastewater treatment, a water reuse system has been designed by the producer and is going to be applied in the future; the foreseen environmental benefits generated by this solution are evaluated in the WR scenario, and results are presented below.

A comparison between the impacts from the production phase (agricultural and processing phase) in the WR and NWR scenarios is presented in Figure 6. The introduction of the water filtering system reduces the impacts for all the categories considered. Such reduction is particularly considerable for ME (-26.5%), CC (-26.6%) and POF (-20.8%). These benefits arise from the reduction of the electricity needed for pumping water and, above all, from the water saving itself, which, in turn, reduces the amount of the wastewater produced.

Nonetheless, the benefits produced by the filtering system could slightly change (namely, being reduced) since, as previously mentioned, the filters used in order to ensure the proper quality of recycled water are not included in the analysis. Taking into account the life cycle of the filters will add a slight extra burden to the environmental performance.

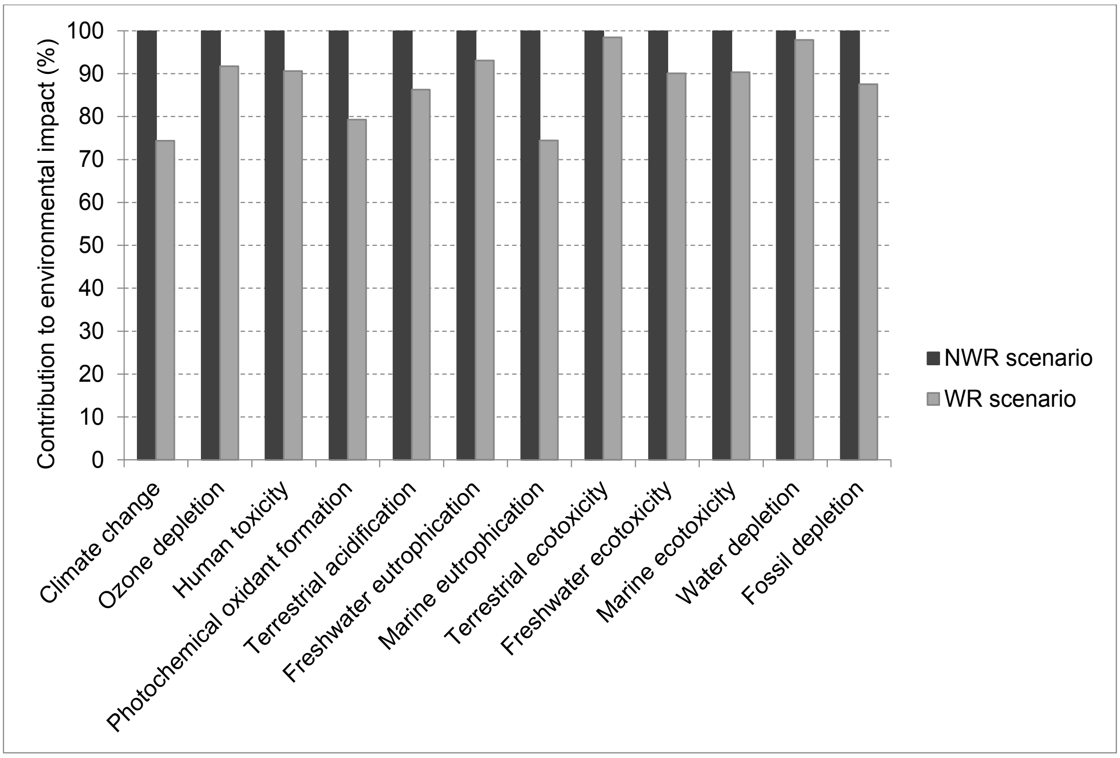


Fig 6 Comparison between the NWR and WR scenarios at the production stage

Currently, no LCA-oriented studies on fresh-cut vegetables are available in the literature. Some authors performed LCA studies on greenhouse crops (although no transformation phase was included), such as tomatoes (Carlsson-Kanayma, 1998; Antón et al., 2005; Roy et al., 2008; Cellura et al., 2012a; Cellura et al., 2012b), zucchinis and peppers (Cellura et al., 2012a; Cellura et al., 2012b). Nonetheless, it is not possible to compare the results not even for the agricultural phase due to the different system boundaries and functional units selected.

As mentioned in 2.6.1, a sensitivity analysis was performed on the NWR scenario. The variation of the IPCC emission factors have large effects on the eutrophication (FE) and acidification (TA) impact categories, which are primarily associated with emissions due to the use of fertilisers. Changes in the emission factors result also in smaller changes for global warming results. The variation of the pesticide emission factors results in changes on the TE category, which is the environmental impact mostly affected by the emissions due to pesticide application.

The sensitivity analysis carried out on allocation showed smaller changes for all the impact categories considered: when the entire environmental burden is allocated only on fresh-cut salad, a maximum increment of 2.73% (and 3.0% if only the production stage is considered) is recorded for ME.

The results of the sensitivity analysis are listed in Table 10.

Table 10. Sensitivity analysis results, calculated for the characterisation step, expressing the changes for each impact category with respect to the reference case (NWR scenario).

Impact categories	N-EMISSION FACTORS VARIATION		PESTICIDES EMISSION FACTORS VARIATION		NO ALLOCATION	
	Variation % on the		*Variation % on the*		*Variation % on the*	
	Agricultural phase	***Whole life cycle***	***Agricultural phase***	***Whole life cycle***	***Production phase***	***Whole life cycle***
Climate Change	-4.52/+19.85	-0.83/+4.36	-	-	+0.10	+0.09
Ozone depletion	-	-	-	-	+0.14	+0.11
Human toxicity	-	-	-	-	+0.12	+0.09
Photochemical oxidant formation	-	-	-	-	+0.13	+0.12
Terrestrial acidification	-36.2/+42	-17.35/+25.8	-	-	+1.68	+1.62
Freshwater eutrophication	-	-	-	-	+0.46	+0.4
Marine eutrophication	-29.7/+37.3	-8.55/+14.6			+3.0	+2.73
Terrestrial ecotoxicity	-	-	-13.9%	-12.2%	+0.22	+0.21
Freshwater ecotoxicity	-	-	-	-	+1.32	+1.06

Marine ecotoxicity	-	-	-	-	+0.24	+0.20
Water depletion	-	-	-	-	+0.002	+0.0015
Fossil depletion	-	-	-	-	+0.07	+0.065

4. Conclusions

This study evaluates the environmental impacts of one bag of fresh-cut lamb's lettuce along its life cycle, namely the agricultural, processing and distribution phases. As already pointed out in other studies carried out on ready-to-eat products, the processing phase generates greater environmental impacts than the agricultural phase. Even though the agricultural activities can be environmentally relevant due to the high use of diesel fuel for agricultural machinery and the use of chemicals as fertilisers and pesticides, the environmental performance of the product analysed is mainly affected by the washing and packaging phases (processing phase) due to the higher consumption of energy and the use of water and related production of wastewater, which need to be treated.

With reference to the use of water and production of wastewater, it should be noted that the consumer who buys a not prepared salad would carry out a washing operation anyway. Nevertheless the way of performing such treatment is different whether it is implemented at home or in a processing plant (e.g., at home a lesser amount of water and energy might be needed). On the contrary, it can also happen that consumers who are more anxious about sanitary concerns also wash at home the already washed and prepared salad.

The agricultural phase represents the second largest contributor to the environmental burden of the salad bag, while the transportation stage appears to be negligible, probably due to the short distance covered for the product's distribution in the case study considered. In addition, contrary to what is commonly thought, the packaging material itself was not found to be relevant.

Possible interventions and solutions to reduce the environmental impacts of the whole supply chain and to optimise the resource efficiency can be addressed both on the consumers' side (i.e., through environmental awareness campaigns about the impacts of the consumption choices, e.g., for ready-to-eat products) or on the production side, i.e., trying to reduce the impacts throughout all the production stages.

As the water consumed during the processing phase is responsible for a large environmental impact, the possibility to install a filtration plant for the recovery of 40% of the washing solution has been evaluated for the case study. The reduction of the environmental impact stemming from the introduction of the water filtering system is considerably relevant for some categories, namely Marine Eutrophication, Climate Change and Photochemical Oxidant Formation.

Based on the results, it could be concluded that the reuse of processing water represents an effective solution for improving the environmental performance of ready-to-eat vegetables, in addition to determining a reduction of the processing costs. As a more general result, the case study considered highlights the environmental burdens associated with the current food production chains, which have become more complex during the last decades. The need to ensure food safety and conservation for longer periods and longer transport distances, and the increasing demand of ready-to-eat food (mainly due to a change in lifestyles) imply the need for longer and more complex production chains. The agricultural phase is only the first step, instead of being a self-standing chain as it was in the recent past.

5 Acknowledgements

This study received financial support from AGER as "STAYFRESH – Novel strategies meeting the needs of the fresh-cut vegetable sector" research project.

6 References

Amienyo D, Gujba H, Stichnothe H, Azapagic A (2013) Life cycle environmental impacts of carbonated soft drinks. Int J Life Cycle Assess 18: 77–92

Andersson K., Ohlsson T, Olsson P (1998) Screening life cycle assessment (LCA) of tomato ketchup: a case study. J Clean Prod 6: 277–288

Antón, A, Montero J I, Muñoz P (2005) LCA and tomato production in Mediterranean greenhouses. International Journal of Agricultural resources Governance and Ecology 4 (2): 102-112

Audsley E (1997) Harmonization of environmental life cycle assessment for agriculture. European Commission DG VI Agriculture. Silsoe Research Institute, Silsoe

Bacenetti J, Negri M, Fiala M, González-García S (2013) Anaerobic digestion of different feedstock: impact on energetic and environmental balances of biogas process Sci Total Environ 463-464: 541-464

Bacenetti J, Fusi A, Negri M, Guidetti R, Fiala M (2014) Environmental assessment of two different crop systems in terms of biomethane potential production. Sci Total Environ 466-467: 1066-77

Benedetto G (2013) The environmental impact of a Sardinian wine by partial Life Cycle Assessment. Wine Economics and Policy 2: 33-41

Berlin J, Sund V (2010) Environmental Life Cycle Assessment (LCA) of ready meals. LCA of two meals; pork and chicken & Screening assessments of six ready meals. SIK Report N° 804 2010

Bevilacqua M, Braglia M, Carmignani G, Zammori F A (2007) Life Cycle Assessment Of Pasta Production In Italy. J Food Quality 30: 932–952

Blengini G, Busto M (2009) The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). J Environ Manage 90: 1512–1522

Milà I Canals L, Chapagain A, Orr S, Chenoweth J, Anton A, Clift R (2010) Assessing freshwater use impacts in LCA, part 2: case study of broccoli production in the UK and Spain. Int J Life Cycle Assess 15: 598-607

Carlsson-Kanayma A (1998) Climate change and dietary choices: how can emissions of greenhouse gases from food consumption be reduced. Food Policy 23: 277-293

CARPI (Consorzio Autonomo Riciclo Plastica Italia) http://www.consorziocarpi.com/. Accessed 18 December 2013

Casati D, Baldi L (2012) Il confezionato tira nonostante la crisi. Terra e Vita 27: 34-37

Castoldi, N., Bechini, L., Ferrante, A., 2011. Fossil energy usage for the production of baby leaves. Energy 36: 86-93

Cellura M, Ardente F, Longo S (2012b) From the LCA of food products to the environmental assessment of protected crops districts: A case-study in the south of Italy. J Environ Manage 93: 194-208

Cellura M, Longo S, Mistretta M (2012a). Life Cycle Assessment (LCA) of protected crops: an Italian case study. J Clean Prod 28: 56-62

Davis J, Sonesson U (2008) Life cycle assessment of integrated food chains — a Swedish case study of two chicken meals. Int J Life Cycle Assess 13: 574–584

Davis J, Wallman M, Sund V, Emanuelsson A, Cederberg C, Sonesson U (2011) Emissions of Greenhouse Gases from Production of Horticultural Products. Analysis of 17 products cultivated in Sweden. SIK, SR 828, ISBN 978-91-7290-301-2, 1-72

DEFRA (2008) Greenhouse Gas Impacts of Food Retailing. Defra Research Project FO 0405. Department for the Environment Food and Rural Affairs, London. http://www.defra.gov.uk/

DETEC (Federal Department of the Environment, Transport, Energy and Communications) http://www.bafu.admin.ch/index.html?lang=it. Accessed 18 December 2013

De Vries JW, Groenestein CM, De Boer IJM (2012) Environmental consequences of processing manure to produce mineral fertiliser and bio-energy. J Environ Manage 102: 173-183

Dressler D, Loewen A, Nelles M (2012) Life cycle assessment of the supply and use of bioenergy: impact of regional factors on biogas production. Int J Life Cycle Assess 17: 1104-1115

EMEP/EEA (2013) Agriculture other (use of pesticides and limestone). In Air pollutant emission inventory guidebook 2013 - Technical report No 12/2013. European Environment Agency: Copenhagen

Ferrante A, Martinetti L, Maggiore, T (2009) Biochemical changes in cut vs. intact lamb's lettuce (*Valerianella olitoria*) leaves during storage. Int J Food Sci Tech 44: 1050-1056

Frischknecht R, Jungbluth N, Althaus HJ, Doka G, Heck T, Hellweg S, Hischier R, Nemecek T, Rebitzer G, Spielmann M, Wernet G (2007) Overview and Methodology. Ecoinvent report No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf

Fusi A, Guidetti R, Benedetto G (2014) Delving into the environmental aspect of a Sardinian white wine: from partial to total Life Cycle Assessment. Sci Total Environ 472: 989-1000

Garnett T (2006) Fruit and Vegetables & UK Greenhouse Gas Emissions: Exploring the Relationship. Working Paper Produced as Part of the Work of the Food Climate Research Network, Centre for Environmental Strategy, University of Surrey

Goedkoop MJ, Heijungs R, Huijbregts M, De Schryver A, Struijs J, Van Zelm R (2009) ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition Report I: Characterisation

González-García S, Bacenetti J, Negri M, Fiala M, Arroja L (2013a). Comparative environmental performance of three different annual energy crops for biogas production in northern Italy. J Clean Prod 43, 71-83

González-García S, Castanheira E G, Dias A C, Arroja L (2013b) Environmental life cycle assessment of a dairy product: the yoghurt. Int J Life Cycle Assess 8: 796–811

Hall G, Rothwell A, Grant T, Isaacs B, Ford L, Dixon J, Kirk M, Friel S (2014) Potential environmental and population health impacts of local urban food systems under climate change: a life cycle analysis case study of lettuce and chicken. Agriculture & Food Security 3:6

Hospido A, Milà I Canals L, McLaren S, Truninger M, Edwards-Jones G, Clift R (2009) The role of seasonality in lettuce consumption: a case study of environmental and social aspects. Int J Life Cycle Assess 14 (5): 381-391

Intergovernmental Panel on Climate Change (IPCC) (2006) IPCC Guidelines for National Greenhouse Gas Inventories Agriculture, Forestry and Other Land Use, Chapter 11: N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application. Vol. 4 IPCC

Maraseni TN, Cockfield G, Maroulis J, Chen G (2010) An assessment of greenhouse gas emissions from the Australian vegetables industry. J Environ Sci Heal B 45 (6): 578-588

Margni M, Rossier D, Crettaz P, Jolliet O (2002) Life cycle impact assessment of pesticides on human health and ecosystems. Agr Ecosyst Environ 93: 379–392

Marsh K., Bugusu B (2007) Food Packaging— Roles, Materials, and Environmental Issues. J Food Sci 72 (3): R39–R55

Martín-Diana A, Rico D, Frías J, Henehan G, Mulcahy J, Barat J, Barry-Ryan C (2006) Effect of calcium lactate and heat-shock on texture in fresh-cut lettuce during storage. J Food Eng 77: 1069 – 1077

Martínez-Blanco J, Muñoz P, Antón A, Rieradevall J (2011) Assessment of tomato Mediterranean production in open-field and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint. J Clean Prod 19: 985-997

Müller H (1997) Determination of the carotenoid content in selected vegetables and fruit by HPLC and photodiode array detection. Zeitschrift für Lebensmitteluntersuchung und-Forschung A 204: 88-94

Olmez H, Kretzschmar U (2009) Potential alternative disinfection methods for organic fresh-cut industry for minimizing water consumption and environmental impact. Lwt-Food Sci Technol 42: 686-693

Pluimers J (2011) An Environmental Systems Analysis of Greenhouse Horticulture in the Netherlands. The Tomato Case. Wageningen University, Netherlands

Poritosh R, Daisuk N, Takahiro O, Qingyi X, Hiroshi O, Nobutaka N, Takeo S (2009) A review of life cycle assessment (LCA) on some food products. J Food Eng 9: 1–10

Ragaert P, Verbeke W, Devlieghere F, & Debevere J (2004). Consumer perception and choice of minimally processed vegetables and packaged fruits. Food Qual Prefer 15(3): 259-270

Rico D, Martin-Diana AB, Barat JM, Barry-Ryan C (2007) Extending and measuring the quality of fresh-cut fruit and vegetables: a review. Trends Food Sci Tech 18(7): 373-386

Romer-Gomez M, Audsley E, Suatez-Rey E (2014) Life Cycle Assessment of cultivating lettuce and escarole in Spain. J Clean Prod 73: 193-203

Roy P, Nei D, Okadome H, Nakamura N, Orikasa T, Shiina T (2008) Life cycle inventory analysis of fresh tomato distribution systems in Japan considering the quality aspect. J Food Eng 86, 225 – 233

Roy P, Nei D, Orikasa T, Xu Q, Okadome H, Nakamura N, Shiina T (2009) A review of life cycle assessment (LCA) on some food products. J Food Eng 90: 1 – 10

The Technical Committee of the International EPD System (2013) Product Category Rules (PCR) – Vegetables (UN CPC 012) http://www.environdec.com/it/. Accessed 14 May 2014

Torrellas M, Antón A, Ruijs M, Victoria N, Stanghellini C, Montero J (2012) Environmental and economic assessment of protected crops in four European scenarios. J Clean Prod 28: 45-55

Venkat K (2012) Comparison of Twelve Organic and Conventional Farming Systems: A Life Cycle Greenhouse Gas Emissions Perspective. J Sustain Agr 36 (6): 620 – 649

Vieux F, Darmon N, Touazi D, Soler LG (2012) Greenhouse gas emissions of self-selected individual diets in France: Changing the diet structure or consuming less? Ecol Econ 75: 91–101

Williams H, Wikström F (2011). Environmental impact of packaging and food losses in a life cycle perspective: a comparative analysis of five food items. J Clean Prod 19 (1): 43–48

Science of the Total Environment 466–467 (2014) 1066–1077

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Environmental assessment of two different crop systems in terms of biomethane potential production

Jacopo Bacenetti, Alessandra Fusi *, Marco Negri, Riccardo Guidetti, Marco Fiala

Department of Agricultural and Environmental Sciences, Production, Landscape, Agroenergy, Università degli Studi di Milano, Via G. Celoria 2, 20133 Milan, Italy

HIGHLIGHTS

- Environmental impact of two crop systems was evaluated
- Biomethane specific production tests were carried out
- Alternative scenarios (different yields and crop management) were assessed
- Maize single crop obtains the better environmental performance
- Critical factors are: fertilizer and diesel fuel emissions and diesel fuel production

ARTICLE INFO

Article history:
Received 22 April 2013
Received in revised form 23 July 2013
Accepted 29 July 2013
Available online 28 August 2013

Editor: Charlotte Poschenrieder

Keywords:
Energy crops
Anaerobic digestion
Life cycle assessment
Environmental sustainability
Biogas

ABSTRACT

The interest in renewable energy sources has gained great importance in Europe due to the need to reduce fossil energy consumption and greenhouse gas emissions, as required by the Renewable Energy Directive (RED) of the European Parliament. The production of energy from energy crops appears to be consistent with RED. The environmental impact related to this kind of energy primarily originates from crop cultivation. This research aimed to evaluate the environmental impact of different crop systems for biomass production: single and double crop. The environmental performances of maize and maize plus wheat were assessed from a life cycle perspective. Two alternative scenarios considering different yields, crop management, and climatic conditions, were also addressed. One normal cubic metre of potential methane was chosen as a functional unit. Methane potential production data were obtained through lab experimental tests. For both of the crop systems, the factors that have the greatest influence on the overall environmental burden are: fertilizer emissions, diesel fuel emissions, diesel fuel production, and pesticide production. Notwithstanding the greater level of methane potential production, the double crop system appears to have the worse environmental performance with respect to its single crop counterpart. This result is due to the bigger quantity of inputs needed for the double crop system. Therefore, the greater amount of biomass (silage) obtained through the double crop system is less than proportional to the environmental burden that results from the bigger quantity of inputs requested for double crop.

1. Introduction

The use of renewable sources for energy production is considered to be a potential solution for reducing the environmental problems derived from fossil fuels (Cherubini et al., 2009; Appels et al., 2011; González-García et al., 2012a). However, the environmental impacts of agricultural production systems have raised concern from national as well as international points of view. According to the Intergovernmental Panel on Climate Change, agriculture contributes a share of 13.5% to global anthropogenic greenhouse gas (GHG) emissions (Bachmaier et al., 2010; Carozzi et al., 2013).

In Europe, the interest in Renewable Energy Sources (RES) has strongly increased due to the need to reduce fossil energy consumption and greenhouse gas (GHG) emissions, as indicated by DIRECTIVE 2009/28/EC (RED) (European Parliament Council, 2010, 2009). According to RED, Italy should be able to produce 17% of primary energy using RES by 2020 (in 2009, energy production from RES was already at 8.86%, and in 2010, it was >10%). In particular, RES should produce 100 TWh/year, covering 26% of electric consumption (Ministero dello Sviluppo Economico, 2012).

Energy crops and corresponding derived bioenergy production are expected to bring environmental, social, and economic benefits. Several studies have reported benefits in terms of the reduction of greenhouse gas emissions, air pollution, acidification, or eutrophication (Brentrup et al., 2004; Buratti and Fantozzi, 2010; Kimming et al., 2011; Bacenetti et al., 2012a; González-García et al., 2012b). However, the

* Corresponding author. Tel.: +39 02 50316873; fax: +39 02 503 16845.
E-mail address: alessandra.fusi@unimi.it (A. Fusi).

0048-9697/$ – see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.scitotenv.2013.07.109

environmental impacts strongly depend on the crop cultivation (Fazio and Monti, 2011; González-García et al., 2012c; Uchida and Hayashi, 2012).

For the achievement of European Union (EU) objectives, the anaerobic digestion (AD) of energy crops and agro-industrial by-products and/or wastes appears to be one of the most promising agro-energy chains (Angelidaki and Ellegaard, 2003; Edelmann et al., 2005; Clemens et al., 2006; Wulfa et al., 2006; Börjesson and Berglund, 2007; Jury et al., 2010; Patterson et al., 2011; Capponi et al., 2012). Among all possible solutions, AD represents one of the most promising ways to use RES (Börjesson and Mattiasson, 2008).

Agricultural raw materials, such as straw and manure, are commonly used for biogas production (Del Prado et al., in press). Livestock activities are in fact widespread in Italy and there is, consequently, a great availability of manure. Nevertheless, the main feeding materials for digesters are often represented by cereal silages (of maize, wheat, and triticale, in particular) (Lansche and Müller, 2012).

In 2010, in Germany which is the country with the most significant biogas production (about 7000 biogas plants), more than one-third of the maize area (2,282,000 ha) was used for bioenergy production (Dressler et al., 2012). In Italy, about 1000 agricultural biogas plants are currently (December 2012) in function (374 located in Lombardy), for a global electrical power of 756 MW (Bacenetti et al., 2013); about 10% of the total maize area (1,172,000 ha) was specifically cultivated for biogas purposes (Casati, 2013).

Although detailed information concerning the silage used for the AD plants is not available, in the areas in which biogas production is more widespread an increase in biomass prices and the value of land has taken place (Povellato, 2011).

The production of cereal ensilage can be carried out mainly with two different crop systems: the single crop system with sorghum or maize FAO classes 600 or 700, or the double crop system with winter cereals (wheat, barley or triticale) followed by maize FAO classes 300, 400 or 500. In Lombardy, the maize hybrids FAO Class 700 are the most used for energy production in the single crop system, while maize hybrids FAO Classes 400 – 500 can be suitable after the harvesting of winter cereals, when the double culture system is chosen. However, the choice between the single crop or the double crop system must be carefully evaluated. In the double crop system, despite a moderated increase of production, field operations as well as production factors used (fertilizers, seeds, pesticide) are approximately double that in maize 700. Consequently, double crop systems involve higher economic costs.

In addition, it must be considered that the possibility for using double crop systems is linked to climatic conditions. The key factor is the speed with which tillage operations and sowing of maize 500 are performed after the harvest of the wheat. In years with a rainy spring this might not be allowed, or it could force the choice of maize hybrids with a shorter vegetative cycle (for example, maize class 300 or 400) instead of maize class 500 (which shows higher biogas production). Environmental effects caused by energy crop cultivation come not only from field operations but also from raw material (fuels, fertilizers, and pesticides) extraction, production and transportation (Scacchi et al., 2010). Therefore, in order to perform a complete evaluation of the system, all of these aspects must be taken into account.

Life cycle assessment (LCA) is a methodology that aims to analyze products, processes, or services from an environmental perspective [ISO 14040, 2006] (Guinée et al., 2002; ISO, 2006), providing a useful and valuable tool for agricultural system evaluation (Audsley, 1997; Brentrup et al., 2001; IPCC, 2006; Finnveden et al., 2009; Fiala and Bacenetti, 2011) as well as for energy crops (Gasol et al., 2009; Bacenetti et al., 2012b; González-García et al., 2012b).

The aim of this study was to analyze the environmental performances of two different crop systems (single and double crop) cultivated in Northern Italy, used for producing biomass for energy purposes. The LCA method was chosen to perform the environmental analysis.

2. Materials

2.1. Goal and scope definition

The environmental performances of a single crop (maize class 700 or maize 700) and a double crop (maize class 500 or maize 500 plus wheat) were compared in terms of methane potential production. Moreover, the most critical stages for both crop systems under study throughout the life cycle were identified.

The choice of the selected biomasses was due to their diffusion in the Lombardy region. Maize 700 is the best maize hybrid for single crop cultivation in the Po Valley area while Maize 500 is the most suitable maize hybrid for second sowing after a winter cereal. Therefore, in these climatic conditions, maize 700 (as a single crop) and the cultivation of wheat followed by maize 500 (as a double crop) are the two solutions which allow the better exploitation of the growing season. For this reason these two cropping systems were evaluated using the LCA methodology.

2.2. Description of the cropping systems under assessment

Wheat (*Triticum* spp. L.), which is a winter crop, and maize (*Zea mais* L.), which is a summer crop, were analyzed. Two FAO maize classes, in particular, were considered: maize 700 and maize 500.

Two different crop systems were taken into account:

1. Single crop: maize 700 only;
2. Double crop: wheat followed by maize 500.

For the double crop system, the seed bed preparation for maize cultivation is realized immediately after harvesting the wheat.

Cultivation of both analyzed crops is located in the Po Valley area, district of Milan, Lombardy region (Italy). The local climate is characterized by an average annual temperature of 12.7 °C, and rainfall is mainly concentrated in autumn and spring (average annual precipitation is equal to 745 mm).

Field operations can be divided into four main steps: (1) soil tillage; (2) crop growth; (3) biomass harvesting and transport; and (4) biomass ensilage. Operations included in each step are shown in Fig. 1. Basically, the two crop systems differ in terms of the land occupation time: 5 and 12 months per year for single and double crop systems, respectively. There are differences between maize and wheat and also between maize 700 and maize 500, with regard to applied fertilizers and pesticides rates, seeds and water, and diesel fuel amounts. Field and ensilage operations for the three crops under study are described in the following subsections and reported as supplementary material.

2.2.1. Maize

Maize is the most widespread summer crop in Italy; in 2011, about 1 million hectares were cultivated [24% in Lombardy] (ISTAT, 2011). In May, before ploughing, the soil will have been fertilized with digestate at rates of 45 and 85 t ha^{-1} for the maize 500 and 700, respectively. After ploughing, always in May, the soil is harrowed, sown, and treated with herbicides [Lumax, 4 kg ha^{-1}]. The sowed seeds range from 70,000 (19 kg ha^{-1}) to 77,000 (20 kg ha^{-1}) seeds ha^{-1} depending on maize classes. In addition to the digestate spreading, mineral fertilization, using potassium- and phosphorous-based fertilizers, is carried out for maize 500 between the ploughing and harrowing. Chemical weed control is carried out twice in June using 1 kg ha^{-1} of Dual. In the same month, hoeing and mineral fertilization with urea are performed.

For maize, irrigation can increase and make the yield steady. After the top fertilization, irrigation is performed between four and five times for maize 500 and 700, respectively, in July and August. The water volume is 800 m^3 ha^{-1} for each intervention. For both of the FAO classes, the harvesting of the maize silage occurs in September with self-propelled machines. After transport to the biogas plant, the biomass is stored in silos, and ensilage is executed for wheat biomass.

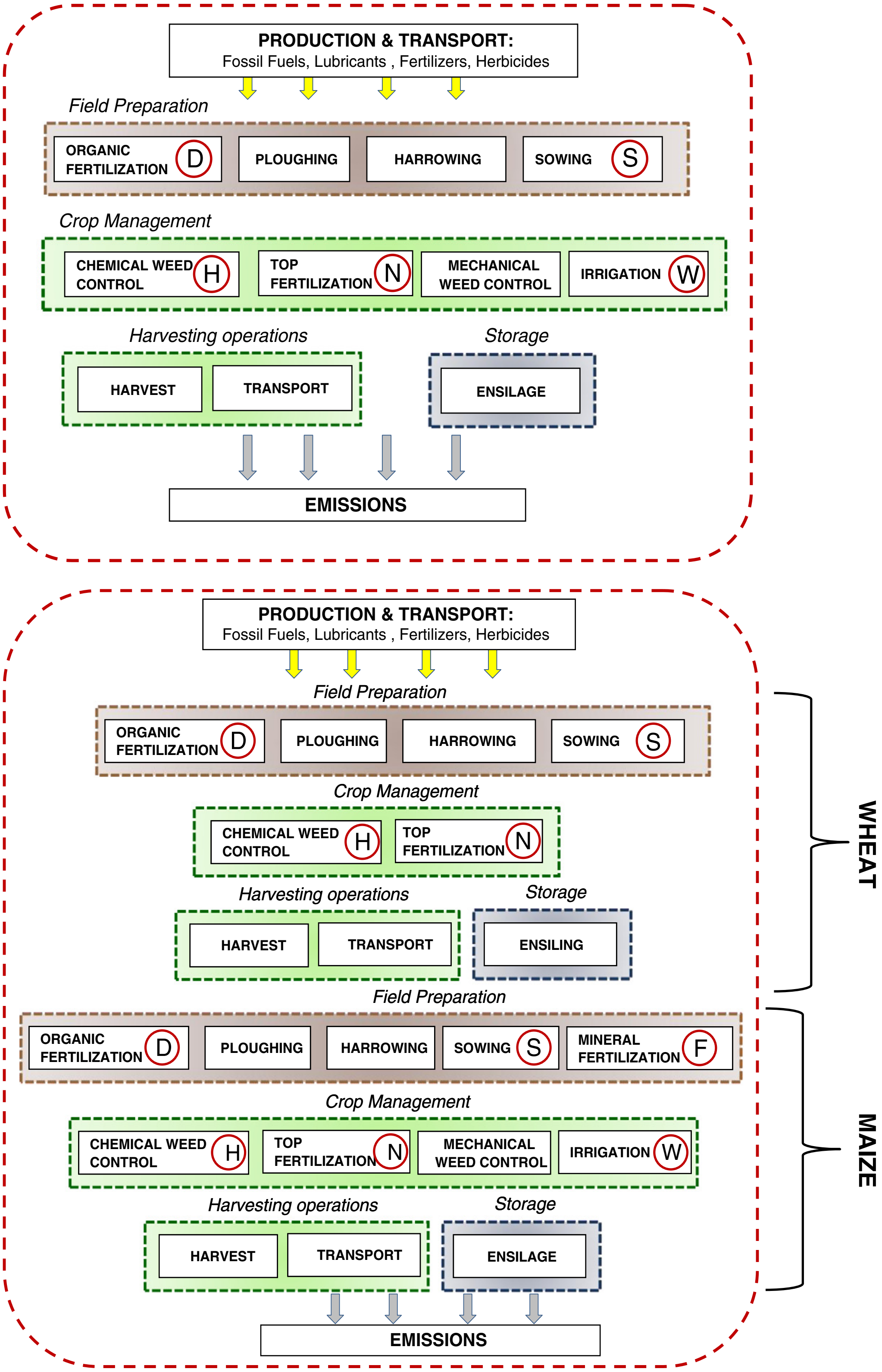


Fig. 1. System boundaries: single crop on the top, double crop on the bottom. Note: D = digestate, S = seeds, H = herbicide, N = nitrogen fertilizer, W = water, F = phosphorous and potassium fertilizers.

Table 1
Methane potential production for the different crops.

Crop system	Crop	Average value
		$m^3{}_N/t_{wb}$
Single crop	Maize 700	102.3 ± 10.9
Double crop	Maize 500	105.6 ± 11.9
	Wheat	83.1 ± 11.8

Large differences exist in the biomass yield and dry matter content between the two different classes of maize: The maize 700 gives a yield that is higher (75 t_{wb} ha^{-1} with a dry matter content of 34%) than that of maize 500 (48.75 t_{wb} ha^{-1} with a dry matter content of 38%).

2.2.2. Wheat

Together with barley and triticale, wheat is one of the most widespread winter crops in Northern Italy [approximately 340,000 ha in 2011] (Casati, 2011). It can be used for human and animal feeding (grain or fodder) and as biomass for AD plants. When employed as an energy crop, wheat is harvested before the grain ripening, and all of the biomass is harvested and chopped in order to be subsequently ensiled.

Before sowing, the soil is first fertilized with digestate (40 t ha^{-1}); after that, the soil is ploughed and harrowed. The digestate comes from nearby AD plants to which the produced biomass is delivered. The spread digestate, which has a dry matter content of 5.4%, is applied during the second half of September, when the average maximum temperature is 26.5 °C and the average minimum temperature is 16.2 °C. No rainfall (1 mm) occurred during the days following the digestate application. The sowing was performed in October using 35,000 seeds ha^{-1} (200 kg ha^{-1}) in order to obtain a final density of 300–350 plants m^{-2}. First, chemical weed control is carried out following the seeding (pre-emergency) by spraying terbutilazine and alachlor (5 kg ha^{-1}). In addition to organic fertilization, chemical fertilizers are also applied in two steps. The first is carried out in November using ammonium nitrate at a rate of 60 kg ha^{-1}; the second is carried out in February with urea (60 kg ha^{-1}). However, mechanical weed control and irrigation are not carried out. In May, the cariosside (seed) reaches the waxy ripeness, and the whole crop (straw and grain) is harvested and chopped using a self-propelled forager that simultaneously loads the biomass into farm trailers that are coupled with tractors and driven beside the foragers. The biomass yield is 38.35 t_{wb} ha^{1} (with a dry matter content of 32%).

The biomass is transported to the biogas plant (distance = 2.5 km). The chopped biomass is ensilaged and stored in horizontal silos for feeding the digesters during the entire year. The ensiling operations are carried out by means of wheeled tractors that are equipped with a frontal loader that compacts and presses the chopped biomass inside the silos.

2.3. Functional unit

The functional unit (FU) selected in many LCA studies of energy crops is the mass of biomass (Dressler et al., 2012; Goglio et al., 2012; González-García et al., 2012a,b). However, the biomasses produced from the energy crops assessed have different characteristics and, consequently, different biogas specific productions. Therefore, the mass of biomass does not appear to be the most appropriate FU because it does not allow a fair comparison among the assessed cereals and crop systems.

Considering that the analysis was performed on crops that were specifically cultivated for energy production by means of AD plants, the selected FU was 1 normal cubic metre (1 $m^3{}_N$) of potential methane (CH_4). Methane potential production data were obtained through experimental tests (Table 1) (described in Section 2.5).

For both crop systems, the annual methane potential production was considered by calculating the volume of potential methane obtained from:

1. 1 ha cultivated with maize 700 (year 2011), and
2. 1 ha cultivated with maize 500 and wheat (year 2011).

Table 2
Field and ensilage operations for single crop (maize 700).

Operation	NN.	Month	Tractor	Operative machine			Note
			Mass Power	Type Size	Mass (kg)	Time (h/ha)	
Pre-seeding organic fertilization	1	May	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	85 t_{wb} ha^{-1} Digestate[a]
Ploughing	1	May	10,500 kg 190 kW	Plough	2000	1.11	–
Harrowing	1	May	7300 kg 130 kW	Rotary Harrow 4.0 m	1800	1.20	–
Sowing	1	May	5050 kg 90 kW	Pneumatic seeder 4 lines	900	1.00	20 kg ha^{-1}
Chemical weeding	3	May Jun Jun	4450 kg 80 kW	Sprayer 15 m	600	0.33	4 kg ha^{-1} lumax 1 kg ha^{-1} dual 1 kg ha^{-1} dual
Irrigation	5	Jun Jul Aug	4450 kg 80 kW	Pump 950 m^3/h	550	1.20	4400 m^3 ha^{-1}
Mechanical weeding	1	Jun	5050 kg 90 kW	Weeder 2.8 m	550	0.33	–
Top fertilization	1	Jun	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} urea
Harvesting	1	Sep	–	Forage harvester 335 kW	13,000	1.00	
Transport	1	Sep	5050 kg 90 kW	3 Farm trailers 30 m^3	5500	3.03	–
Ensilage	1	Set	5050 kg 90 kW	2 Frontal loader 2 m^3	450	3.03	

[a] Average composition: N = 0.40%; P_2O_5 = 0.08%, K_2O = 0.31%.

2.4. System boundaries

The studied system (Fig. 1) included crop cultivation and harvesting, biomass transport, and ensilage to the biogas plant. For the two crop systems, the life cycle of each agricultural process was included within the system boundaries. This life cycle considers raw materials extraction (e.g. fossil fuels and minerals), manufacture (e.g. seeds, fertilizers, chemicals and agricultural machines) and use (diesel fuel consumption and derived combustion). The agronomic inputs for the two crop systems under assessment are shown in Tables 2 and 3, where the characteristics of agricultural machines commonly used for these crops are summarized.

Three different scenarios were considered:

1. The baseline scenario (BS) represents the situation as it was recorded and described within Tables 4 and 5; the average value was considered.
2. Alternative scenario 1 (AS1), or "hypothetical future scenario," is a scenario in which an increase of 15% of biomass yield (and, subsequently, of CH4 production) and fertilizer application was assumed. This scenario considers favorable climatic conditions in addition to proper plant nutrition as well as the development of improved maize hybrids (for example, genetically modified organisms with resistance to pests and/or to drought). The impact of yield increase on environmental performances has already been evaluated in several LCA studies (for example. González-García et al., 2012a,b). Although several authors have studied the possibility of getting a yield raise as a consequence of an increase in fertilizer application (Bélanger et al., 2012; El-Fouly et al., 2012; Gagnon et al., 2012; Latković et al., 2012; Tremblay et al., 2012) the increase of fertilizer application has been hypothesized only in order to keep balanced the ratio between nitrogen application and nitrogen removal.
3. Alternative scenario 2 (AS2), or "worst case scenario," is a scenario in which all agricultural inputs were kept constant, and a decrease in yield of 15% was hypothesized as a consequence of adverse weather conditions (for example, hailstorms, strong drought) or due to inadequate phytosanitary management (for example, unexpected/late detection of *Ostrinia nubilalis* and or *Diabrotica virgifera virgifera* attacks).

2.5. Data collection

Data (from 2011) concerning field operations, ensilage, and transport were directly obtained via questionnaires that were administered to farmers and via surveys on the field. The farmer provided all

Table 3
Field and ensilage operations for double cropping (wheat + maize 500).

	Operation	NN.	Month	Tractor	Operative machine			Note
				Mass Power	Type Size	Mass (kg)	Time (h/ha)	
WHEAT	Pre-seeding organic fertilization	1	Sep	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	40 t_{wb} ha^{-1} Digestate[a]
	Ploughing	1	Sep	10,500 kg 190 kW	Plough 3-shovel	2000	1.11	
	Harrowing	1	Sep	7300 kg 130 kW	Rotary harrow 4.0 m	1800	1.20	
	Seeding	1	Oct	5050 kg 90 kW	Seeder	900	1.00	200 kg ha^{-1}
	Mechanical Weeding	1	Oct	4450 kg 80 kW	Spraying 15 m	600	0.33	Terbutilazina + Alachlor 5 kg ha^{-1}
	Top fertilization	2	Nov Feb	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} ammonium nitrate 60 kg ha^{-1} urea
	Harvesting	1	May	–	Forage harvester 335 kW	13,000	1.00	
	Transport	1	May	5050 kg 90 kW	2 Farm trailers 30 m^3	5500	2.00	
	Ensilage	1	May	5050 kg 90 kW	2 Frontal loader 2 m^3	450	2.00	
MAIZE 500	Pre-seeding organic fertilization	1	May	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	45 t ha^{-1} Digestate
	Ploughing	1	May	10,500 kg 190 kW	Plough 3-shovel	2000	1.11	–
	Post-seeding mineral fertilization	1	May	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	100 kg ha^{-1} P_2O_5 and K_2O
	Harrowing	1	May	7300 kg 130 kW	Rotary harrow 4,0 m	1800	1.20	–
	Seeding	1	May	5050 kg 90 kW	Pneumatic seeder 4 lines	900	1.00	19 kg ha^{-1}
	Chemical weeding	3	May Jun Jun	4450 kg 80 kW	Sprayer 15 m	600	0.33	1 kg ha^{-1} dual 4 kg ha^{-1} lumax
	Irrigation	4	Jun, 2 Jul Aug	4450 kg 80 kW	Pump 950 m^3/h	550	1.20	3600 m^3 ha^{-1}
	Weeding	1	Jun	5050 kg 90 kW	Weeder 2.8 m	550	0.33	
	Top fertilization	1	Jun	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} urea
	Harvesting	1	Sep	–	Forage Harvester 335 kW	13,000	1.00	
	Transport	1	Sep	5050 kg 90 kW	3 Farm trailers 30 m^3	5500	3.03	
	Ensilage	1	May	5050 kg 90 kW	2 Frontal loader 2 m^3	450	3.03	

[a] Average composition: N = 0.40%; P_2O_5 = 0.08%, K_2O = 0.31%.

Table 4
Data inventory for single crop (data related to FU: 1 m^3_N of potential CH_4).

Input from the technosphere	Units	
Fertilizers		
Digestate[a]	kg	10.8
Urea as N	g	3.51
Pesticides		
Metolachlor S	mg	412.9
Triazine compounds	mg	95.29
Mesotrione	mg	19.06
Fuel		
Diesel fuel for field operations and transport	g	26.4
Diesel fuel for ensilage	g	4.53
Other inputs		
Lubricants	g	0.63
Maize seeds	g	2.54
Input from the environment		
Land use	m^2	1.27
Irrigation water	m^3	0.56
Output to the environment		
Product		
Maize 700 silage[b]	kg	9.78
Emissions to the atmosphere		
Ammonia (fertilizer)	g	7.33
Nitrous oxide (fertilizer)	g	0.49
HC (diesel)	mg	64.8
CO (diesel)	g	0.34
CO_2 (diesel)	g	87.9
NO_x (diesel)	g	0.69
Particulate (diesel)	mg	44.06
Metolachlor S	mg	9.62
Mesotrione	mg	0.01
Triazine compounds	mg	0.10
Emissions to water		
Phosphate	mg	86.39
NO_3	g	2.46
Metolachlor S	mg	0.36
Mesotrione	mg	0.19
Triazine compounds	mg	7.34

[a] = By-product of AD: it does not contribute to environmental impact; [b] = Losses (5% of harvested biomass) due to the transport and the ensilage operation were taken into consideration (Bacenetti et al., 2013).

Table 5
Data inventory for double crop (data related to FU: 1 m^3_N of potential CH_4).

Input from the technosphere	Units	
Fertilizers		
Digestate[a]	kg	10.73
Urea as N	g	6.97
Fertilizer P_2O_5	g	12.63
Fertilizer K_2O	g	12.63
Pesticides		
Metolachlor S	mg	284.15
MCPA	mg	101.03
Triazine compounds	mg	94.72
Clopyralid	mg	10.10
Fluroxypyr	mg	20.21
Mesotrione	mg	18.94
Fuel		
Diesel fuel for field operations & transport	g	38.07
Diesel fuel for ensilage	g	4.84
Other inputs		
Lubricants	g	0.88
Maize seeds	g	2.40
Wheat seeds	g	25.26
Input from the environment		
Land use	m^2	1.26
Irrigation water	m^3	0.45
Output to the environment		
Products		
Maize 500 silage[b]	kg	5.87
Wheat silage	kg	4.57
Emissions to the atmosphere		
Ammonia (fertilizer)	g	7.82
Nitrous oxide (fertilizer)	g	0.56
HC (diesel)	mg	90.84
CO (diesel)	g	0.48
CO_2 (diesel)	g	124.85
NO_x (diesel)	g	0.97
Particulate (diesel)	mg	69.02
Metolachlor S	mg	6.91
Mesotrione	mg	0.01
MCPA	mg	3.84
Triazine compounds	mg	0.10
Clopyralid	μg	23.24
Fluroxypyr	mg	0.36
Emissions to water		
Phosphate	mg	212.2
Metolachlor S	mg	0.26
MCPA	mg	2.77
Mesotrione	mg	0.18
Triazine compounds	mg	6.63
Clopyralid	mg	0.31
Fluroxypyr	mg	44.66

[a] = By-product of AD: it does not contribute to environmental impact; [b] = Losses (5% of harvested biomass) due to the transport and the ensilage operation were taken into consideration (Bacenetti et al., 2013).

information regarding fertilizer, pesticide (registered in a mandatory document called "Quaderni di campagna"), and water use. The diesel fuel consumption was partly measured (by evaluating the volume of fuel used to fill up fuel tanks to the brim) and partly estimated using the model SE^3A (Fiala and Bacenetti, 2012).

Emissions due to the fertilizers were included: nitrogen emissions (nitrate, ammonia, and nitrous oxide) were computed following the model proposed by Brentrup et al. (2000), while phosphate emissions were calculated in accordance with Smil (2000). Climatic data for 2011, which were necessary for calculating fertilizer emissions, were obtained from the meteorological station closest to the farm.

Pesticide emissions were also estimated using PestLCI (Birkved and Hauschild, 2006), a model that quantifies the emissions to different environmental compartments (i.e. groundwater, surface water, and air). Regarding fuel use, the emissions that each machine in field operations generated were estimated using data from the Swiss Federal Office for the Environment (Federal Department of the Environment, Transport, Energy and Communications, or DETEC) (DETEC, 2013). Background data for seed production, diesel fuel, fertilizers, and pesticides were obtained from the Ecoinvent database [version 2.2] (Frischknecht et al., 2007) and the LCA Food DK database (Nielsen et al., 2003).

No change in the overall soil carbon content has been assumed because the fields were previously dedicated to cereal cultivation (González-García et al., 2012a,b). Tables 4 and 5 report data inventories for the single and double crop under study.

Data concerning CH_4 potential production were obtained by means of laboratory experimental tests. For the different biomasses, the specific productions were obtained using lab-scale AD tests; unstirred lad-fermenters (volume: 2.5 dm^3) were utilized. These were made up of a hermetically sealed glass jar equipped with a metallic cover containing a valve through which the biogas produced reaches the corresponding gasometer. Each gasometer is made from a methacrylate pipe (volume 3.5 dm^3) atop which are fitted two hoses: One transports the biogas from the fermenter, and one, equipped with a valve, can be used for gasometer recharge with the water solution (saturated with $CaCO_3$ to

Table 6
Evaluated impact categories (CML 2000).

Impact categories	Unit of measure
Ozone layer depletion (ODP)	kg CFC-11equivalents
Human toxicity	1.4-dichlorobenzene equivalents
Fresh water aquatic ecotoxicity	1.4-dichlorobenzene equivalents
Marine aquatic ecotoxicity	1.4-dichlorobenzene equivalents
Terrestrial ecotoxicity	1.4-dichlorobenzene equivalents
Photochemical oxidation	kg ethylene equivalents
Global warming (GWP100)	kg carbon dioxide equivalents
Acidification	kg SO_2 equivalents
Abiotic depletion	kg antimony equivalents
Eutrophication	kg PO_4 equivalents

prevent CO_2 solubilisation into water) of which it is full at the beginning of the measurement.

Samples of fermenting material from different full-scale AD were collected to be used as inoculum. Before fermenters were set up, they were filtered with 2 mm sieves and placed at 40 °C for 48 h to minimize the amount of inoculum biogas production.

In each fermenter, the inoculum/substrate ratio was kept at 2:1 on a volatile solids basis (Vismara et al., 2008): On average, each fermenter contained 2 kg of inoculum (total solids 3% ± 0.2 of raw material) and 30 g of dried biomass. Before digestion, all substrates were ground. During the experimental tests, the temperature in each fermenter was 40 °C. To keep the biomass conditions as homogeneous as possible

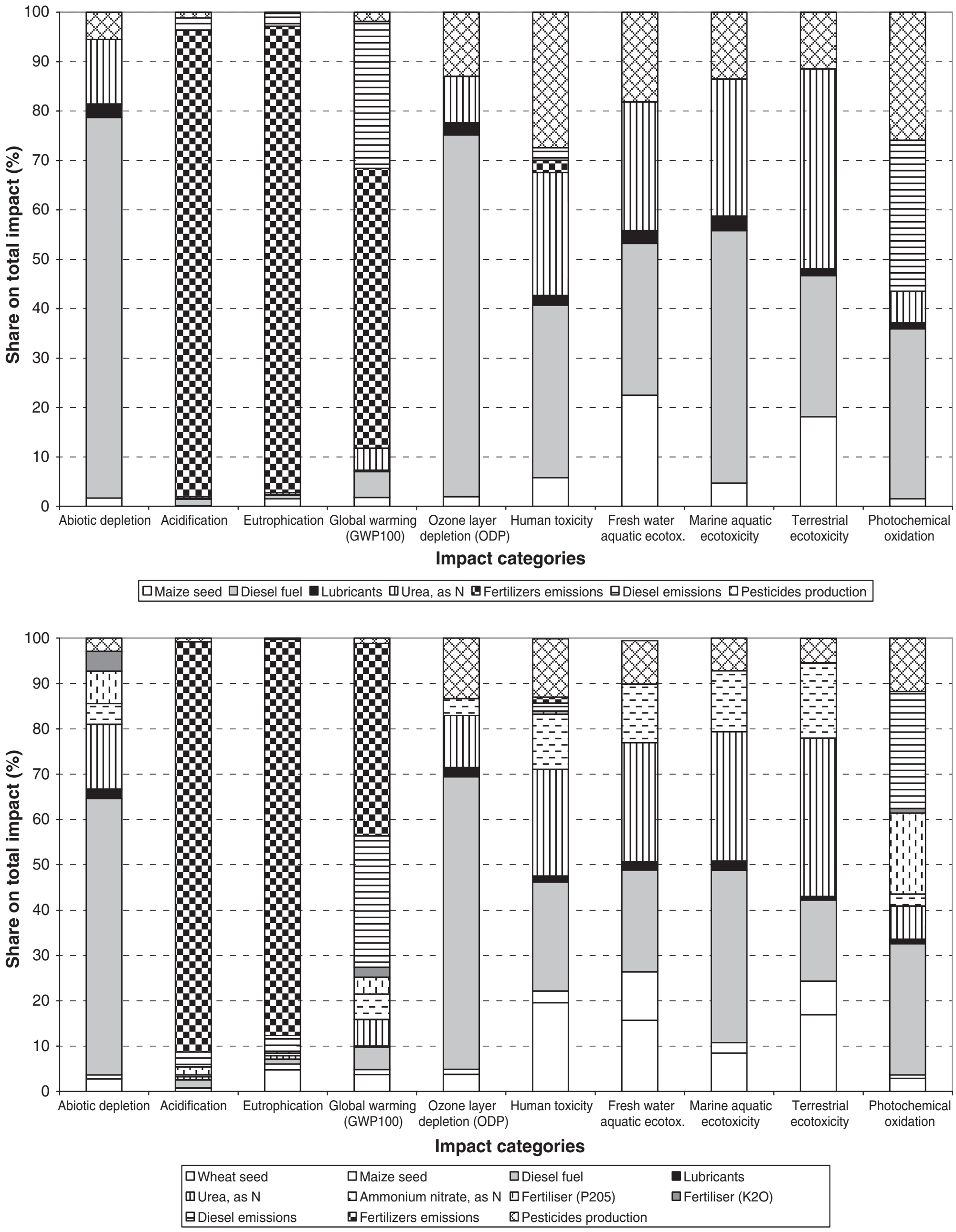


Fig. 2. Environmental impact of 1 m^3_N of CH_4: from maize 700 (top) and from wheat + maize 500 (bottom).

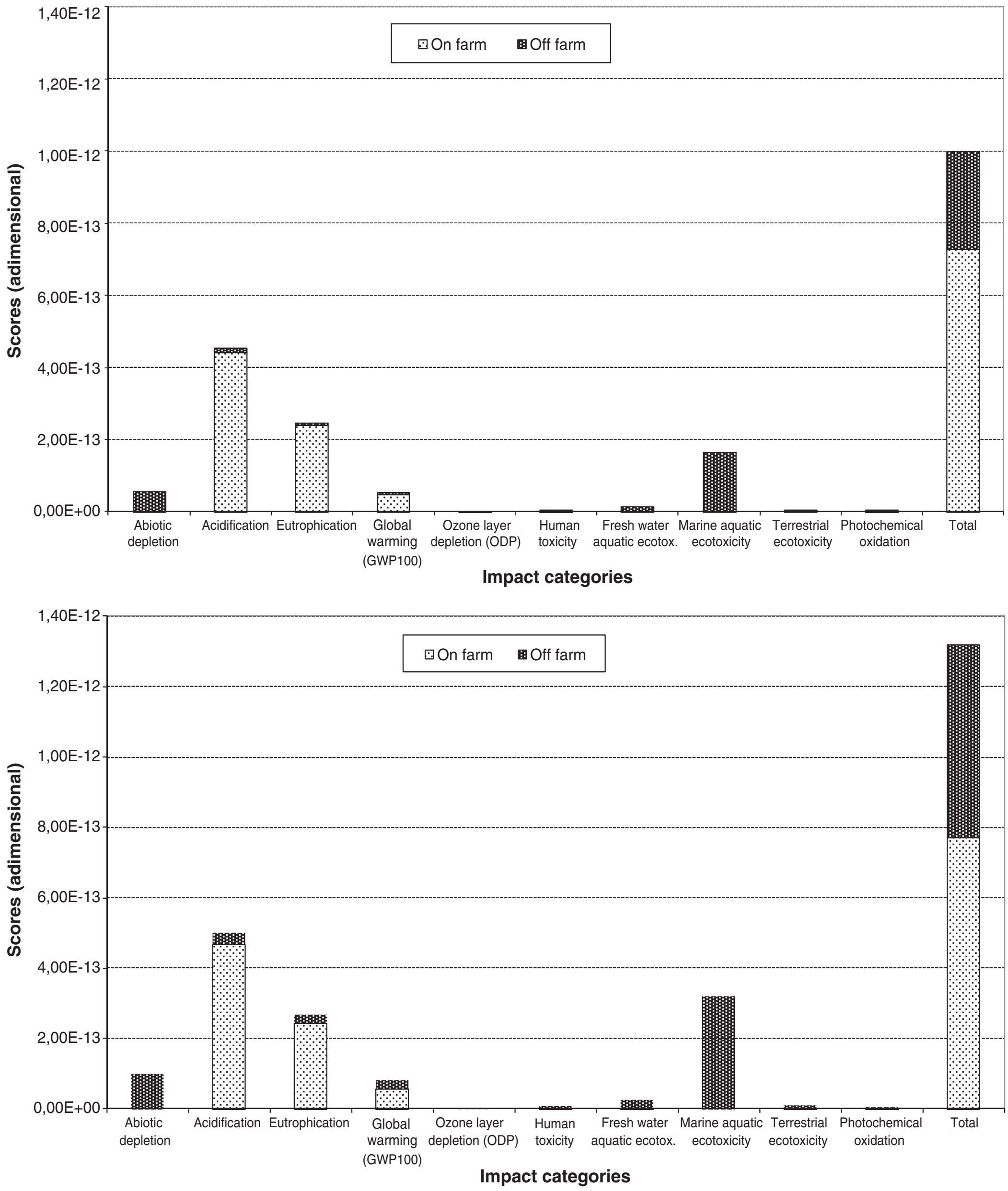


Fig. 3. Environmental impact (subdivided into "on-farm" and "off-farm") to obtain 1 m^3_N of CH_4 from the cultivation of maize 700 (top) and of wheat + maize 500 (bottom).

and to facilitate biogas collection, the fermenters were shaken daily. Biogas production was recorded daily by reading the centimeters run in the gasometers and by calculating the equivalent volume.

Before AD, the dry matter total solid (TS) content was determined following a 24-h drying period at 105 °C, while volatile solid (VS) content was determined as a percentage of TS, according to APHA (1998). Biogas composition in terms of CH_4 percentage was monitored by means of one "Binder Combigas GA-m^3" (from Binder, D) portable gas analyzer.

2.6. Methods

Among the steps defined within the life cycle impact assessment stage of the standardized LCA methodology, classification, characterization and normalization stages were carried out in this study. The characterization factors reported by the Centre of Environmental Science of Leiden University (CML 2001 method) were used (Guinée et al., 2002). The impact categories evaluated according to the CML method are reported in Table 6. A life cycle impact assessment (LCIA) was performed using SimaPro software (Goedkoop et al., 2008). LCIA was performed for the three different scenarios as previously discussed.

3. Results and discussion

The environmental impact linked to CH_4 production from maize 700 is widely determined, as expected, by maize cultivation. Ensilage plays a secondary role in the environmental burden for all of the three crops

Table 7
Comparison between 1 m^3_N obtained from maize 700 (single crop) and from wheat + maize500 (double crop). The worst crop system was set at 100.

Impact categories	Single crop	Double crop
	%	%
Abiotic depletion	57.70	100
Acidification	90.70	100
Eutrophication	91.83	100
Global warming (GWP100)	66.94	100
Ozone layer depletion (ODP)	63.59	100
Human toxicity	49.13	100
Fresh water aquatic ecotox.	51.80	100
Marine aquatic ecotoxicity	53.51	100
Terrestrial ecotoxicity	44.46	100
Photochemical oxidation	60.20	100

Table 8
Values for the impact categories.

Impact category	Units	Methane potential from maize 700	Methane potential from maize 500 + wheat
Abiotic depletion	kg Sb eq	9.26E − 04	1.61E − 03
Acidification	kg SO_2 eq	1.25E − 02	1.38E − 02
Eutrophication	kg PO_4 eq	3.12E − 03	3.36E − 03
Global warming (GWP100)	kg CO_2 eq	2.71E − 01	4.10E − 01
Ozone layer depletion (ODP)	kg CFC-11 eq	1.87E − 08	2.95E − 08
Human toxicity	kg 1.4-DB eq	3.14E − 02	6.26E − 02
Fresh water aquatic ecotox.	kg 1.4-DB eq	7.39E − 03	1.43E − 02
Marine aquatic ecotoxicity	kg 1.4-DB eq	2.04E + 01	3.82E + 01
Terrestrial ecotoxicity	kg 1.4-DB eq	2.10E − 04	4.73E − 04
Photochemical oxidation	kg C_2H_4 eq	2.91E − 05	4.84E − 05

evaluated. Considering its importance in terms of environmental impact, the cultivation phase is the stage of the life cycle that is worth attention. Fig. 2 shows, for the two crop systems, the contributions to the impact categories of each input and output for obtaining 1 m^3_N of CH_4.

The critical factors are: fertilizer emissions (decisive for acidification, eutrophication and global warming potential), diesel fuel emissions (important for global warming potential and photochemical oxidation), diesel fuel production (decisive for abiotic and ozone layer depletion), pesticide production (important for human toxicity), and urea production. In more detail, for both the crop systems analyzed, approximately 90% of eutrophication and acidification derived from fertilizer emissions. Global warming potential originating from fertilizer emissions was over 50% for the single crop and 40% for the double crop. The abiotic and ozone layer depletion impacts are primarily caused by diesel fuel consumption in both cases. Diesel fuel emissions are responsible for almost 30% of GWP and photochemical oxidation impacts. The contribution of urea production ranged from 25% (human and water toxicity) to 30% (double crop) and 40% (single crop) (terrestrial ecotoxicity). Pesticides production was relevant primarily for the single crop; its contribution ranged from 12% (terrestrial ecotoxicity) to almost 30% (human toxicity).

Normalized data[1], shown in Fig. 3, were subdivided into "on-farm" and "off-farm" impacts. On-farm impacts represent the environmental burden derived directly from farm activities (such as diesel fuel emissions and fertilizers emissions); meanwhile, off-farm impacts are not directly related to farm activities (inputs production).

For the single crop system, the overall environmental burden mainly stems from on-farm impacts. For the double crop, the overall burden is almost equally due to both on-farm and off-farm impacts. For both the single and double crop, the most relevant impact categories were acidification, eutrophication, marine aquatic ecotoxicity and abiotic depletion. The first two were mainly caused by farm activities, the latter two were the result of off-farm activities.

Comparing the cultivation phases for the single (maize 700) and double (maize 500 + wheat) crop, it can be stated that for each impact category, the cultivation of maize 700 (to obtain 1 m^3 of methane) in the single crop system is environmentally more sustainable than it is in the double crop system.

The comparison in terms of the environmental impact of 1 m^3_N of CH_4 obtained from maize 700 and from maize 500 plus wheat turns out to be very favourable for maize 700 (Table 7). The environmental burden of the double crop is greater than that of the single crop for each impact category; for acidification and eutrophication, the differences are less evident (<10%). Knowing that fertilizer emissions largely influence these impact categories, the main reason for such a result is N leaching. Notwithstanding that for the double crop more fertilizer is used, no leaching occurs (according to a calculation from Brentrup et al., 2000). On the contrary, cultivation of maize 700, although it requires a lower fertilizer application, determines nitrate leaching, which causes acidification and eutrophication. Table 8 shows scores for all impact categories for methane from the single and double crop systems.

Considering the maximal and minimal CH_4 production (see Table 1), the overall environmental impact grows as CH_4 production decreases. Concerning the comparison among different scenarios (BS, AS1, AS2) the following can be considered. For maize 700 (Fig. 4), the reduction of the yields (AS2) (keeping all other conditions constant except for the diesel fuel used for ensilage) leads to a proportional decrease of CH_4 production, causing an aggravation of the environmental burden of approximately the same entity than the yield decrease (15%). On the contrary, in AS1, all environmental impacts decrease (−20% approximately) except for acidification and eutrophication, which remain essentially the same.

For maize 500 plus wheat (Fig. 4), comparing BS with AS2, the same considerations could be made except that eutrophication varies more than the yield reduction does. In AS2, the lower yield leads to nitrate leaching (absent in BS), which exacerbates the disparity for the eutrophication impact category.

Considering that it is essential that the production of feedstock is carried out under sustainable conditions, in recent years, several LCA studies have been carried out to evaluate the environmental impact of energy crops cultivation (Dressler et al., 2012; Goglio et al., 2012; González-García et al., 2012b; Bachmaier et al., 2013; Ghahderijani et al., 2013). Our results are in agreement with this literature. In fact, although the use of a different FU does not allow for a strict comparison, all these LCA studies highlighted that the process hotspots are: i) nitrogen fertilization (which involves remarkable impacts due to its production as well as its application into the soil); ii) diesel fuel consumption (mainly for ploughing and harvest, field operations with high power requirements); and iii) pesticide utilization for impact categories such as human toxicity and terrestrial ecotoxicity. In more detail, regarding fertilizer emissions Dressler et al. (2012) and González-García et al. (2012b) also reported a strong relation between organic fertilization and eutrophication.

4. Conclusions

This study compares the environmental performances of a single and double crop system. The evaluation has been made using 1 m^3_N of methane as a functional unit due to the final use of the crops considered.

[1] The CML method includes data normalization: the results for each impact category were divided by a reference. This reference is the average inhabitant environmental load (for each impact category) in Europe in 1995. The normalization step allows the obtaining of adimensional scores.

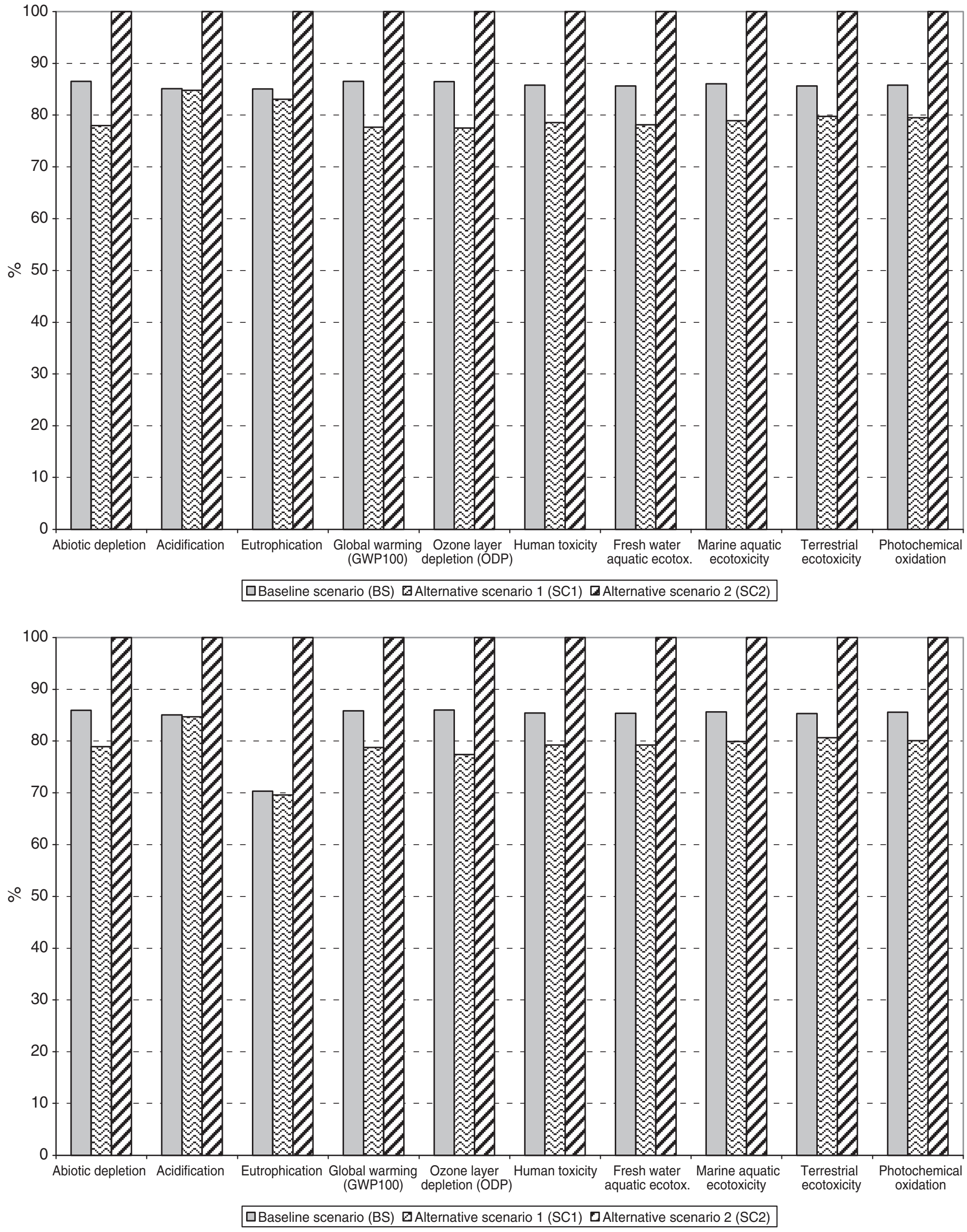


Fig. 4. Comparison among different scenarios for maize 700 (top) and wheat + maize 500 (bottom).

Results from the analysis point out that for all impact categories taken into consideration, the double crop system appears to have the worse environmental performance compared to the single crop system. This means that the greater amount of biomass (silage) obtained with the double crop system is less than proportional to the environmental burden that results from the bigger quantity of inputs needed for the double crop system.

Comparing scenarios that differ for biomass yields, it is evident that the more the yield increases, the more the environmental burden decreases. The same supposition can be drawn for specific methane production: The more the methane specific production increases, the better the environmental performance of the system.

The analysis executed highlighted that the nitrogen cycle and their linked emissions are relevant for the environmental burden of maize and wheat cultivation, especially for some impact categories (namely acidification and eutrophication). Therefore, the model chosen to estimate nitrogen emissions in the environment is a critical factor in this kind of analysis due to its influence on the final result. For this reason, the analysis of this aspect of cultivation should be, if possible, performed with site-specific models.

Because the biomass produced in the two crop systems under study is used to feed anaerobic digestion plants, the next step of our study will be the analysis of the conversion phase of the biomass into biogas and then into electricity. The results of the current analysis represent the first essential step for the whole life cycle assessment of electricity production from AD realized in agricultural plants.

Acknowledgments

The present work is funded by Regione Lombardia - Fondo per la Promozione di Accordi Istituzionali, project BIOGESTECA 15083/RCC". In addition, the Authors thank Regione Lombardia which financed a Postdoctoral Research Fellowship ("Progetto Dote Ricerca" financed by FSE - Regione Lombardia).

References

Angelidaki I, Ellegaard L. Codigestion of manure and organic wastes in centralized biogas plants: status and future trends. Appl Biochem Biotechnol 2003;109:95–105.
Appels L, Lauwers J, Degrève J, Helsen L, Lievens B, Willems K, et al. Anaerobic digestion in global bio-energy production: potential and research challenges. Renew Sustain Energy Rev 2011;15:4295–301.
Audsley E. Harmonisation of environmental life cycle assessment for agriculture. Final Report, Concerted Action AIR3–CT94–2028. European Commission, DG VI Agriculture; 1997. p. 1–139.
Bacenetti J, González-García S, Mena A, Fiala M. Life cycle assessment: an application to poplar for energy cultivated in Italy. J Agric Eng 2012a;43:72–8.
Bacenetti J, Mena A, Negri M, Cantarella P, Bocchi S, Fiala M. Energetic and environmental balance of a biogas plant in Northern Italy. Proceedings CIGR-AgEng International Conference on Agricultural EngineeringValencia. CIGR-AgEng; 2012b.
Bacenetti J, Negri M, Fiala M, Gonzalez Garcia S. Anaerobic digestion of different feedstock: impact on energetic and environmental balances of biogas process. Sci Total Environ 2013;463–464:541–51.
Bachmaier J, Effenberger M, Gronauer A. Greenhouse gas balance and resource demand of biogas plants in agriculture. Eng Life Sci 2010;10:560–9.
Bachmaier H, Effenberger M, Gronauer A, Boxberger J. Changes in greenhouse gas balance and resource demand of biogas plants in southern Germany after a period of three years. Waste Manag Res 2013;31(4):368–75.
Bélanger G, Claessens A, Ziadi N. Grain N and P relationships in maize. Field Crop Res 2012;126:1–7.
Birkved M, Hauschild M. PestLCI-a model for estimating field emissions of pesticides in agricultural LCA. Ecol Model 2006;198:433–51.
Börjesson P, Berglund M. Environmental systems analysis of biogas systems – Part II: The environmental impact of replacing various reference systems. Biomass Bioenergy 2007;31:326–44.
Börjesson P, Mattiasson B. Biogas as a resource-efficient vehicle fuel. Trends Biotechnol 2008;26:7–13.
Brentrup F, Küsters J, Lammel J, Kuhlmann H. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int J Life Cycle Assess 2000;5:349–57.
Brentrup F, Küsters J, Kuhlmann H, Lammel J. Application of the life cycle assessment methodology to agricultural production: an example of sugar beet production with different forms of nitrogen fertilisers. Eur J Agron 2001;14:221–33.
Brentrup F, Küsters J, Lammel J, Barraclough P, Kuhlmann H. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur J Agron 2004;20:265–79.
Buratti C, Fantozzi F. Life cycle assessment of biomass production: development of a methodology to improve the environmental indicators and testing with fiber sorghum energy crop. Biomass Bioenergy 2010;34:1513–22.
Capponi S, Fazio S, Barbanti L. CO_2 savings affect the break-even distance of feedstock supply and digestate placement in biogas production. Renew Energy 2012;37:45–52.
Carozzi M, Ferrara RM, Rana G, Acutis M. Evaluation of mitigation strategies to reduce ammonia losses from slurry fertilisation on arable lands. Sci Total Environ 2013;449:126–33.
Casati D. Le superfici possono ricominciare a salire. Terra Vita 2011;35:40–6.
Casati D. Annata davvero difficile urge risalire la china. Terra Vita 2013;6:40–4.
Cherubini F, Bargigli S, Ulgiati S. Life cycle assessment (LCA) of waste management strategies: landfilling, sorting plant and incineration. Energy 2009;34:2116–23.
Clemens J, Trimborn M, Weiland P, Amon B. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agr Ecosyst Environ 2006;112(2–3):171–7.
Del Prado A, Mas K, Pardo G, Gallejones P. Modelling the interactions between C and N farm balances and GHG emissions from confinement dairy farms in northern Spain. Sci Total Environ 2013;465:156–65.
DETEC(Federal Department of the Environment, Transport, Energy and Communications). http://www.bafu.admin.ch/index.html?lang=it. [last visit 05/02/2013].
Dressler D, Loewen A, Nelles M. Life cycle assessment of the supply and use of bioenergy: impact of regional factors on biogas production. Int J Life Cycle Assess 2012;17(9):1104–15.
Edelmann W, Baier U, Engeli H. Environmental aspects of the anaerobic digestion of the organic fraction of municipal solid wastes and of solid agricultural wastes. Water Sci Technol 2005;52:203–8.
El-Fouly M, Abou El-Nour E, Shaaban S, Zeidan M. Effect of different levels of NPK and micronutrients fertilization on yield and nutrient uptake of maize plants. J Am Sci 2012;8(8):209–14.
European Parliament Council. Report from the commission to the council and the European Parliament on sustainability requirements for the use of solid and gaseous biomass sources in electricity, heating and cooling; 2010.
European parliament and council. Directive 2009/28/EC on the promotion of the use of energy from renewable sources, vol. L 283. Official Journal of the European Communities; 2009. p. 33.
Fazio S, Monti A. Life cycle assessment of different bioenergy production systems including perennial and annual crops. Biomass Bioenergy 2011;35:4868–78.
Fiala M, Bacenetti J. Economic, energetic and environmental impact in Short Rotation Coppice harvesting operations. Biomass Bioenergy 2011;42:107–13.
Fiala M, Bacenetti J. Model for the economic, energetic and environmental evaluation in biomass productions. J Agric Eng 2012;42:26–35.
Finnveden G, Hauschild M, Ekvall T, Guine J, Heijungs R, Hellwege S, et al. Recent developments in Life Cycle Assessment. J Environ Manage 2009;91:1–21.
Frischknecht R, Jungbluth N, Althaus HJ, Doka G, Heck T, Hellweg S, et al. Overview and methodology. Ecoinvent report No. 1. Dübendorf: Swiss Centre for Life Cycle Inventories; 2007.
Gagnon B, Ziadi N, Grant C. Urea fertilizer forms affect grain corn yield and nitrogen use efficiency. Can J Soil Sci 2012;92(2):341–51.
Gasol CM, Gabarrell X, Anton A, Rigola M, Carrasco J, Ciria P, et al. LCA of poplar bioenergy system compared with Brassica carinata energy crop and natural gas in regional scenario. Biomass Bioenergy 2009;33:119–29.
Ghahderijani M, Komleh S, Keyhani A, Sefeedpari P. Energy analysis and life cycle assessment of wheat production in Iran. Afr J Agric Res 2013;8(18):1929–39.
Goedkoop M, de Schryver A, Oele M. Introduction to LCA with SimaPro 7. the Netherlands: PRé Consultants; 2008.
Goglio P, Bonari E, Mazzoncini M. LCA of cropping systems with different external input levels for energetic purposes. Biomass Bioenergy 2012;42:33–42.
González-García S, Bacenetti J, Murphy R, Fiala M. Present and future environmental impact of poplar cultivation in Po valley (Italy) under different crop management systems. J Clean Prod 2012a;26:56–66.
González-García S, Bacenetti J, Negri M, Fiala M, Arroja L. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J Clean Prod 2012b;43:71–83.
González-García S, Mola-Yudego B, Dimitriou I, Aronsson P, Murphy R. Environmental assessment of energy production based on long term commercial willow plantations in Sweden. Sci Total Environ 2012c;421–422:210–9.
Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, Koning A, et al. Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Kluwer. Dordrecht: Academic Publishers; 2002. p. 692.
IPCC. Agriculture, forestry and other land use. In: Eggleston, et al, editor. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme; 2006.
ISO. Environmental management-life cycle assessment-principles and framework (ISO 14040). Brussels: European Committee for Standardization; 2006.
ISTAT - Istituto Nazionale di Statistica. Tavola C02 - Superficie (ettari) e produzione (quintali): riso, mais, sorgo, altri cereali. Dettaglio per regione - Anno. http://agri.istat.it/jsp/dwExcel.jsp?q=plC020000010000012000&an=2011&ig=1&ct=244&id=18A|15A|25A, 2011. [last access 21/12/12].
Jury C, Benetto E, Koster D, Schmitt B, Welfring J. Life Cycle Assessment of biogas production by monofermentation of energy crops and injection into the natural gas grid. Biomass Bioenergy 2010;34:54–66.
Kimming M, Sundberg C, Nordberg Å, Baky A, Bernesson S, Norén O, et al. Biomass from agriculture in small-scale combined heat and power plants – a comparative life cycle assessment. Biomass Bioenergy 2011;35:1572–81.
Lansche J, Müller J. Life cycle assessment of energy generation of biogas fed combined heat and power plants: environmental impact of different agricultural substrates. Eng Life Sci 2012;12:313–20.
Latković D, Marinković D, Crnobarac J, Jaćimović G, Berenji J, Sikora J. Influence of crop residues and increasing rates of nitrogen on the yield of corn. Res J Agric Sci 2012;44(3):61–6.
Ministero dello Sviluppo Economico. Decreto ministeriale 6 luglio 2012 – Incentivi per energia da fonti rinnovabili elettriche non fotovoltaiche. http://www.sviluppoeconomico.gov.it/images/stories/normativa/DM_6_luglio_2012_sf.pdf, 2012. [last access 24/12/2012].
Nielsen PH, Nielsen AM, Weidema BP, Dalgaard R, Halberg N. LCA food data base. www.lcafood.dk, 2003.
Patterson T, Esteves S, Dinsdale R, Guwy A. Life cycle assessment of biogas infrastructure options on a regional scale. Bioresour Technol 2011;15:7313–23.
Povellato A. Aumenta la superficie in affitto delle aziende professionali. Inf Agrar 2011;40:32–9.
Scacchi C, González-García S, Caserini S, Rigamonti L. Greenhouse gases emissions and energy use of wheat grain-based bioethanol fuel blends. Sci Total Environ 2010;408(21):5010–8.
Smil V. Phosphorus in the environment: natural flows and human interferences. Annu Rev Energy Environ 2000;25:53–88.

Tremblay N, Bouroubi Y, Bélec C, Mullen R, Kitchen N, Thomason W, et al. Corn response to nitrogen is influenced by soil texture and weather. Agron J 2012;104(6):1658–71.
Uchida S, Hayashi K. Comparative life cycle assessment of improved and conventional cultivation practices for energy crops in Japan. Biomass Bioenergy 2012;36:302–15.
Vismara R, Malpei F, Centemero M. Biogas da rifiuti solidi urbani. Palermo: Flaccovio Editore; 2008.
Wulfa S, Jägerb P, Döhlerb H. Balancing of greenhouse gas emissions and economic efficiency for biogas-production through anaerobic co-fermentation of slurry with organic waste. Agr Ecosyst Environ 2006;112:178–85.

ARTICLE IN PRESS

Journal of Cleaner Production xxx (2014) 1–11

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Impact of cropping system and soil tillage on environmental performance of cereal silage productions

Jacopo Bacenetti*, [1], Alessandra Fusi [1], Marco Negri, Marco Fiala

Department of Agricultural and Environmental Sciences, Production, Landscape, Agroenergy, Università degli Studi di Milano, Via G. Celoria 2, 20133 Milano, Italy

ARTICLE INFO

Article history:
Received 12 May 2014
Received in revised form
30 July 2014
Accepted 18 August 2014
Available online xxx

Keywords:
Life Cycle Assessment
Functional unit
Minimum tillage
No tillage
Cropping systems
Silage

ABSTRACT

In this study the environmental performances of the two most widespread cropping systems for cereal silage production in Northern Italy were analysed. Three different technical solutions for the seedbed preparation (conventional tillage, minimum tillage, and no tillage) were considered too. The Life Cycle Assessment method was chosen for the environmental analysis. The following impact potentials were evaluated: abiotic depletion, climate change, ozone depletion, acidification, eutrophication, and photochemical oxidant formation. One ton of dry matter was chosen as the functional unit. Taking into account that the functional unit selection can affect the environmental results, a sensitivity analysis was performed considering three other different functional units (area, biomethane production, and nutritive value).

For both the crop systems, the emissions due to fertiliser application, diesel fuel consumption and production are the hotspots process with the greater influences on the overall environmental burden. Compared to single crop, the double crop system shows the worst environmental performance for all the evaluated impact categories except for euthrophication and acidification (−21% and −14%, respectively). Among the different technical solutions for seedbed preparations, the minimum tillage and the sod seeding achieve better results than the conventional tillage. For impact categories such as abiotic depletion, photochemical oxidation, climate change and ozone layer depletion there are impact reductions ranging from −2.5% to −11.5% for single crop and from −9.4% to −11.7% for double crop. For acidification and eutrophication the impact reduction is minimal for single crop while, for minimum tillage in double crops a slight increase is observed.

1. Introduction

The food system, including agricultural steps as well as transport, processing, and disposal, is one of the main industries responsible for anthropogenic greenhouse gases (GHG) emissions. In 2008, the GHG emissions from this sector were 9800–16,900 Mt of carbon dioxide equivalent ($MtCO_2eq$) (Gerber et al., 2013).

Among the different subsystems of the food system, agriculture has had the greatest impacts, contributing 7300 to 12,700 $MtCO_2eq$/year (including indirect emissions associated with land-cover change), which is equivalent to 80–86% of total food systems' emissions. Inside the agriculture sector, the main source of GHG emissions are deforestation and land use change (30–50% of agricultural emissions) while other activities like soil tillage, crops cultivation, and livestock represent about 70–50% of agricultural emissions (Gerber et al., 2013). Therefore, over the years, the need to assess the environmental impacts of agriculture has become increasingly important (González-García et al., 2012a; Poeschl et al., 2012b; Lijó et al., 2014a).

Among the different crops, annual and perennial, the cereals have an important role in terms of the cultivated area, and they constitute a very important component of the economy as well (FAO, 2013). In Italy, the cereals cultivation covered 3.59 millions of ha, about 28% of the total agricultural area (ISTAT, 2010). Nevertheless, the production of cereal crops involves environmental, social, and economic issues (Poeschl et al., 2012a; Cherubini et al., 2009; Lijó et al., 2014b). Over the years several studies highlighted that the environmental impact of cereal crops can be remarkable (Iriarte et al., 2010; Uchida and Hayashi, 2012; Bacenetti et al., 2012).

Environmental effects due to cereals cultivation (e.g., climate change, acidification, eutrophication, etc.) stem not only from field

* Corresponding author.
E-mail address: jacopo.bacenetti@unimi.it (J. Bacenetti).
[1] These authors equally contributed to this study.

http://dx.doi.org/10.1016/j.jclepro.2014.08.052
0959-6526/

operations (Deytieux et al., 2012) but also from materials (fuels, fertilisers, and pesticides) extraction, processing, and transport (Capponi et al., 2012; Bacenetti et al., 2013). In more details, soil tillage operations, primary (with plough, ripper) and secondary (with harrows, hoe), involve high fuel consumption and consequently have a deep impact on environmental burdens of agricultural processes (in particular, in impact categories such as global warming potential and abiotic resources depletion) (Castanheira and Freire, 2013; Kustermann et al., 2013; Bacenetti et al., 2013b). Alternative solutions for seedbed preparation such as minimum tillage and no tillage have been evaluated in several studies (Santilocchi and Bianchelli, 2006; Basso et al., 2011; Kennedy et al., 2013). Nevertheless, the performances of these solutions have been addressed mainly from an economic and energetic point of view (Basso et al., 2011; Zhang et al., 2013; Santin-Montanya et al., 2013). Carozzi et al. (2013a) assessed the productive, economic and environmental performances of the maize cropping system in conventional agriculture, minimum tillage and no tillage in the Po Valley area.

Nowadays, in Northern Italy, the most widespread cropping systems to produce silages are:

i) The single crop (SC) system, in which only one crop is cultivated season after season. In this type of cropping system, sorghum and (mainly) maize (hybrids with cultivation cycles longer than 105 days, e.g., FAO's classes 600 and 700) are the most cultivated crops.
ii) The double crop (DC) system, in which two crops grow in the same field in sequence. Usually, in this cropping system, a winter cereal (wheat or triticale) is followed by maize (hybrids with cultivation cycles shorter than 105 days, e.g., FAO class 300–400–500). Between wheat and triticale, this second cereal is the most used for silage production in northern Italy due to higher biomass yield (Giunta and Motzo, 2004; Bechini and Castoldi, 2009) its higher specific biogas production (Negri et al., 2014a);

In regards to the environmental impact, although with the DC system a moderately higher dry matter production per hectare can be obtained (Carrosio, 2013; Negri et al., 2014b), the choice between SC and DC must be carefully evaluated. In the DC system, despite a moderated increase of yields, the field operations and the input consumption (fertilisers, seeds, pesticides, water, fuels) are approximately twice that of the SC system.

In the last decade, in order to evaluate the environmental performances of agricultural processes, the Life Cycle Assessment (LCA) has become more and more employed. LCA is a methodology that aims to analyse products, processes, or services from an environmental perspective (Guinée, 2002; ISO, 2006), providing a useful and valuable tool for agricultural system evaluation (Audsley, 1997; IPCC, 2006; Mangena and Brent, 2006)).

Therefore, the aim of this paper was to analyse the environmental performances of the most widespread cropping systems for silage production in Northern Italy, taking into account different technical solutions for seedbed preparation. Although the environmental impact of cereal crops has been already evaluated (Gonzalez et al., 2012b), in this study the attention is not paid to the crop but the cropping system (understood as a sequence of crops grown in the same field during the year). In more detail, the environmental performances of the SC system (only maize 700) were compared with those of the DC system (winter cereals followed by maize). Regarding the seedbed preparation, two alternative scenarios were analysed: minimum tillage and no tillage.

2. Materials and methods

2.1. Goal and scope definition

The goal of this study is to assess the environmental performances of two cropping systems of cereal crops for silage production. The selected cereal crops are the most cultivated in the Po Valley (one of the most important Italian areas for cereal production) (ISTAT, 2012). The silages produced from these cereal crops are utilized mainly for animal feeding (mainly cattle and pigs) but the cereals can also be used for starch production as well as for as human food or for biogas production (Carrosio, 2013; Lijó et al., 2014a, 2014b).

The target audience of this study are the farmers' associations and local politicians.

In this study, maize FAO 700, maize FAO 500, and triticale cultivations were evaluated using the LCA methodology considering two different cropping systems (SC and DC) as well as different practices for seedbed preparation. In more details, SC was cultivated in 2012 while the DC took place in 2011 and 2012. In regards to soil tillage and sowing, three different technical solutions were analysed:

1. The baseline scenario (BS) represents the situation as it was recorded and described within Tables 1 and 2. Ploughing is carried out at 35 cm depth for maize and at 30 cm depth for triticale while harrowing is performed at 15 cm depth for both the crops. Sowing takes place using a traditional seed drill.
2. Minimum Tillage (MT) is an alternative scenario in which soil tillage is different from the BS: the ploughing is replaced by soil tillage (with ripper of 20 cm depth) and the harrowing is carried out with a disc harrow (10 cm depth). Sowing is similar to the BS.
3. Sod seeding (SS) (also called no tillage) is another alternative scenario in which

sowing directly takes place on not-tilled soil. The no tillage technique requires specialized seeding equipment designed to sow into soil covered with crop residues; a double-disk, no-till seed-drill was considered for sowing.

As for the biomass production values and the mechanization of field operations for seedbed preparation, the results obtained by Basso et al. (2011) and by Santilocchi and Bianchelli (2006) were used in this study since they analysed different soil tillage managements in the same geographic area and for soils having similar characteristics.

2.2. Functional unit

The functional unit is an important step of any life cycle assessment since it provides the reference to which all other data in the assessment are normalized. With LCA's application to agricultural processes, different functional units (FUs) can be selected. In many LCA studies of agricultural production systems, the FU is the area (e.g., 1 ha) (Mila i Canalis et al., 2006; Negri et al., 2014). Nevertheless, the mass-based functional unit is prevalent in LCA studies of agricultural systems (Van der Werf et al., 2007; Gonzalez et al., 2012a). Therefore, in this study, 1 t of dry matter produced in each cropping systems has been considered as the FU.

2.3. System boundaries and cropping system description

Cultivations are carried out in the Po Valley area, the Lombardy region (Italy), the District of Pavia, and, more precisely, on the

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Table 1
Field and ensilage operations for SC System (maize FAO Class 700).

Operation	N.	Month	Tractor	Operative machine			Note
			Mass Power	Type Size	Mass (kg)	Working time (h/ha)	
Pre-seeding organic fertilization	1	May	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	85 t_{wb} ha^{-1} Digestate[a]
Ploughing	1	May	10,500 kg 190 kW	Plough	2000	1.11	–
Harrowing	1	May	7300 kg 130 kW	Rotary harrow 4.0 m	1800	1.20	–
Sowing	1	May	5050 kg 90 kW	Pneumatic seed drill 4 lines	900	1.00	20 kg ha^{-1}
Chemical weeding	3	May Jun Jun	4450 kg 80 kW	Sprayer 15 m	600	0.33	4 kg ha^{-1} lumax 1 kg ha^{-1} dual 1 kg ha^{-1} dual
Irrigation	5	Jun Jul Aug	4450 kg 80 kW	Pump 950 m^3/h	550	1.20	4400 m^3 ha^{-1}
Hoeing	1	Jun	5050 kg 90 kW	Hoeing machine 2.8 m	550	0.33	–
Top fertilization	1	Jun	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} urea
Harvesting	1	Sep	–	Forage harvester 335 kW	13,000	1.00	
Transport	1	Sep	5050 kg 90 kW	3 Farm trailers 30 m^3	5500	3.03	–
Ensilage	1	Sep	5050 kg 90 kW	2 Frontal loader 2 m^3	450	3.03	

[a] Average composition of digestate: N = 0.40% P_2O_5 = 0.08% K_2O = 0.31% (Bacenetti et al., 2013).

experimental farm of the University of Milan, which is located in the city of Landriano (45°19′00″N; 9°16′00″E) and ain a nearby farm. The local climate is characterized by an average annual temperature of 12.7 °C, and the rainfall is mainly concentrated in autumn and spring (average annual precipitation is equal to 745 mm). The average texture of the soil of the two farms is 51% sand, 32% silt, and 15% clay.

The life cycle of each agricultural process has been included within the system boundaries (cradle-to-farm perspective). Therefore, this life cycle considers raw materials extraction (e.g., fossil fuels and minerals), manufacture (e.g., seeds, fertilisers, water and agricultural machines), use (diesel fuel consumption and derived combustion and tyre abrasion emissions), maintenance and final disposal of machines, and supply of inputs to the farm (e.g., fertilisers and herbicides). Due to the high level of mechanization of the farm, the indirect environmental burdens of capital goods (tractors, operative machines and buildings) were also included.

Regarding the cereals cultivation in Po Valley, it must be underlined that, especially in areas with irrigation, the crop cycle is characterized by a standardized sequence of field operations. Therefore, from one year to another, the main differences regard mainly the timing in which these operations are performed. For this reason, the crop cultivation analysed in this study is similar to the one reported by other studies (Gonzalez-Garcia, 2012b; Bacenetti et al., 2013, 2014; Negri et al., 2014a; Borrelli et al., 2014).

Triticale cultivation requires about 240–250 days (from the end of September to the first half of June), whereas maize cultivation only requires about 100 days for class 500 (from mid-June to the second half of September) or about 150 days for class 700 (from early May to late September). Fig. 1 reports the sequence of crops in the two cropping systems; in DC the maize Class 500 is sown immediately after the harvest of triticale without uncultivated periods; instead, in SC the soil not cultivated between two growing seasons of maize Class 700.

Fig. 2 represents all the stages and corresponding processes involved. In more details, the cultivation practices for the different cereals (field and ensilage operations) are reported in Tables 1 and 2. Field operations can be divided into (1) soil tillage, (2) sowing, (3) crop growth, (4) biomass harvesting and transport, and (5) biomass ensilage. The crop cultivation starts with organic fertilization and ends with harvesting operation and ensiling.

The biomass yields for the three cereals crops are 77.01 t_{wb} ha^{-1} (wet basis, 67% moisture) for maize class 700, 51.11 t_{wb} ha^{-1} (wet basis, 65% moisture) for maize class 500, and 39.90 t_{wb} ha^{1} (wet basis, 62% moisture) for triticale.

A detailed description of the different agricultural operations for the cereal crops under study is reported by Gonzalez-Garcia et al. (2012b). However, in this study, besides the different geographic areas in which cultivation takes place, the main differences in the cultivation practice previously described (Gonzalez Garcia et al., 2012b) refer to the irrigation technique (in this case, carried out by means of pumps coupled to tractors with a remarkable fuel consumption) and to the biomass transport from field to farm (this process is now included in the system boundaries).

2.4. Inventory data acquisition

Data concerning the field operations, ensilage, and transport were directly obtained by means of questionnaires to farmers as well surveys on the fields. The two cropping systems (DC and SC) were carried out at two different farms located in Landriano (District of Pavia – Northern Italy). More specifically, the DC system was carried out at the "Cascina Marianna," the experimental farm of over 16.5 ha of the Faculty of Agricultural Sciences (University of Milan); while the SC system was carried out on 9.7 ha in a nearby cereal farm located in the same Municipality.

Information regarding seeds, fertilisers, herbicides, and water use were provided by the farmers. This information is registered in a mandatory document called "farmer bookkeeper". The diesel fuel consumption was partly measured (by evaluating the volume of fuel used to fill up fuel tanks to the brim) and partly estimated (for

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Table 2
Field and ensilage operations for DC system (triticale + maize FAO Class 500).

Operation	NN.	Month	Tractor	Operative machine			Note
			Mass, Power	Type, Size	Mass (kg)	Time (h/ha)	
TRITICALE							
Pre-seeding organic fertilization	1	Sep	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	40 t_{wb} ha^{-1} Digestate[a]
Ploughing	1	Sep	10500 kg 190 kW	Plough 3-shovel	2000	1.11	
Harrowing	1	Sep	7300 kg 130 kW	Rotary harrow 4.0 m	1800	1.20	
Seeding	1	Oct	5050 kg 90 kW	Mechanical seed-drill	900	1.00	200 kg ha^{-1}
Chemical Weeding	1	Oct	4450 kg 80 kW	Spraying 15 m	600	0.33	Terbutilazina + Alachlor 5 kg ha^{-1}
Top fertilization	2	Nov Feb	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} ammonium nitrate 60 kg ha^{-1} urea
Harvesting	1	Jun	–	Forage harvester 335 kW	13,000	1.00	
Transport	1	Jun	5050 kg 90 kW	2 Farm trailers 30 m^3	5500	2.00	
Ensilage	1	Jun	5050 kg 90 kW	2 Frontal loader 2 m^3	450	2.00	
MAIZE FAO Class 500							
Pre-seeding organic fertilization	1	Jun	5050 kg 90 kW	Manure spreader 20 m^3	2000	3.33	45 t ha^{-1} Digestate[a]
Ploughing	1	Jun	10,500 kg 190 kW	Plough 3-shovel	2000	1.11	–
Post-seeding mineral fertilization	1	Jun	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	100 kg ha^{-1} P_2O_5 and K_2O
Harrowing	1	Jun	7300 kg 130 kW	Rotary harrow 4.0 m	1800	1.20	–
Seeding	1	Jun	5050 kg 90 kW	Pneumatic seed-drill 4 lines	900	1.00	19 kg ha^{-1}
Chemical Weeding	3	May Jun Jun	4450 kg 80 kW	Sprayer 15 m	600	0.33	1 kg ha^{-1} dual 4 kg ha^{-1} lumax
Irrigation	4	Jun, 2 Jul Aug	4450 kg 80 kW	Pump 950 m^3/h	550	1.20	3600 m^3 ha^{-1}
Hoeing	1	Jun	5050 kg 90 kW	Hoeing machine 2.8 m	550	0.33	
Top fertilization	1	Jun	6850 kg 120 kW	Fertilizer spreader 2500 dm^3	500	0.13	60 kg ha^{-1} urea
Harvesting	1	Sep	–	Forage Harvester 335 kW	13.000	1.00	
Transport	1	Sep	5050 kg 90 kW	3 Farm trailers 30 m^3	5500	3.03	
Ensilage	1	Sep	5050 kg 90 kW	2 Frontal loader 2 m^3	450	3.03	

[a] Average digestate composition: N = 0.40%; P_2O_5 = 0.08%; K_2O = 0.31% (Bacenetti et al., 2013a,b).

harrowing and top fertilization) using the model SE^3A (Fiala and Bacenetti, 2012). The agricultural processes reported in the database Ecoinvent (v. 2.2) have been modified[2] considering the characteristics (mass, power, life span, specific fuel consumptions, etc.) of the machines (tractors and implements) used in the farms involved in the analysis and taking into account the fuel consumptions recorded by means of surveys, farmer's interviews and experimental measures.

The emissions due to diesel fuel use were estimated using the Swiss Federal Office for the Environment Database (Federal Department of the Environment, Transport, Energy and Communications, or DETEC). Secondary data for seeds production, diesel fuel, fertilisers, and herbicides were obtained from the Ecoinvent database (Frischknecht et al., 2007).

The biomass yields were measured by means of the farms' weighbridge.

Emissions due to the fertilisers include nitrogen emissions (nitrate, ammonia, and nitrous oxide) were computed according to Brentrup et al. (2000) and Brentrup et al. (2001) considering climatic data (e.g., temperature, wind, rainfall) and soil conditions (e.g., pH, texture, cation exchange capacity). Climatic data for the year 2011, necessary for calculating fertilisers emissions, were obtained from the meteorological station closest to the experimental farm. Phosphate emissions were calculated following Smil (2000); in more details, a percentage of P loss equal to 1.5% was considered.

Pesticides emissions were estimated using PestLCI (Birkved and Hauschild, 2006).

[2] The Ecoinvent database takes into account, for every agricultural process: the diesel fuel consumption and the amount of agricultural machinery and of the shed, the amount of emissions to the air from combustion and the emission to the soil from tyre abrasion during the work process. The fuel consumption and the associated emissions were varied by inserting the experimental data; the same approach was adopted for the mass of the machines.

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

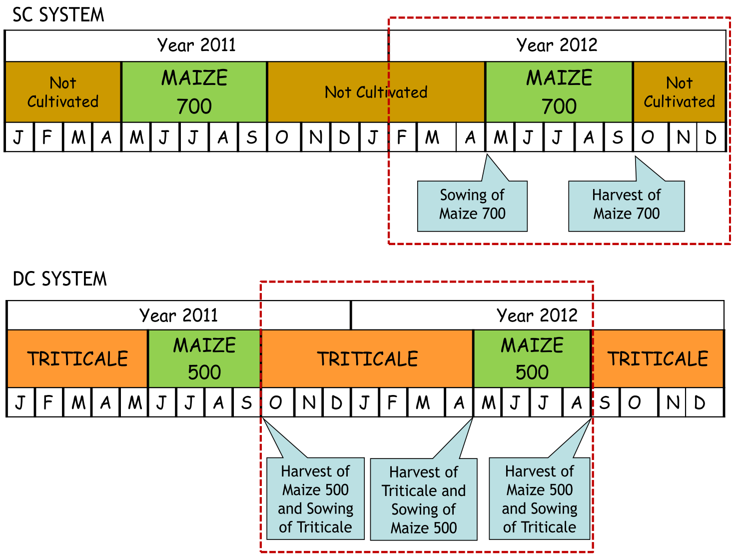


Fig. 1. Land occupation: Comparison between single (SC) and double (DC) cropping systems. The red dashed line indicates the time period considered in the analysis. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Considering that the fields were previously dedicated to cereals cultivation, zero changes in the overall soil carbon content were considered (González-García et al., 2012a; Capponi et al., 2012; Bacenetti et al., 2014).

In regards to the three scenarios, Table 3 summarizes the difference regarding soil tillage and sowing as well as the variation of the diesel fuel consumption.

2.5. Impact assessment

A life cycle impact assessment (LCIA) was performed using SimaPro 7.3.3 software (PRé Consultants) (Goedkoop et al., 2010), and CML 2000 (Guinée, 2002) was chosen to assess the environmental impact. The following impact potentials were evaluated according to the selected method: abiotic depletion (AD), climate change (CC), ozone depletion (OD), acidification (AP), eutrophication (EP), and photochemical oxidant formation (POF).

3. Results

Environmental impact is widely determined by field operations. These operations are mainly responsible for the environmental burdens of the 6 impact categories; on average, less than 5% of the overall environmental impact is due to ensilage.

Fig. 3 shows that for both cropping systems analysed, among the different inputs and outputs, the key aspects of the environmental impact are fertilisers emissions (mainly for acidification and eutrophication), diesel fuel emissions (mainly for global warming potential), and diesel fuel production (mainly for abiotic depletion and ozone layer depletion). The emissions related to fertiliser application (e.g., nitrogen leaching, ammonia volatilization, phosphorous erosion, etc.) are mainly responsible for acidification (with a relative contribution of 94% and 90% for SC and DC, respectively) and eutrophication (relative contribution of 95% and 87% for SC and DC, respectively). Diesel fuel production is the main hotspot in AD (81% and 66% for SC and DC, respectively), OD (77% and 69% for SC and DC, respectively), and POF (40% and 33% for SC and DC, respectively). Emissions due to the combustion of diesel fuel in the machine engines play an important role for CC (29% and 28% for SC and DC, respectively) and POF (28% and 23% for SC and DC, respectively). Herbicide production is critical only for POF (24% and 11% for SC and DC, respectively) and OD (11% and 12% for SC and DC, respectively). Fertiliser production is relevant for AD (11% and 27% for SC and DC, respectively) and for CC (4% and 17% for SC and DC, respectively); these differences are due to the higher fertiliser applications in the DC system. Finally, for all the evaluated impact categories, seed production has a minor importance (less than 2% in SC and less than 6% in DC) as well as lubricant oil production (less than 2% in SC and DC).

Table 4 reports the scores for all impact categories, referred to the selected FU.

The SC system presents better environmental performances than the DC system for 4 of the 6 evaluated impact categories. In particular, the DC shows a considerably higher (about +25%) environmental impact for abiotic depletion and photochemical oxidation. With regard to acidification and eutrophication, the environmental burden of the DC is lower than the SC (about − 15% and −20%, respectively).

4. Discussion

4.1. Comparison with previous studies

Although the environmental burdens of silage productions were evaluated in several studies, a comparison of the results is difficult because different methodology (Bechini and Castoldi, 2009; Carozzi et al., 2013a), characterization methods, system boundaries (Gonzalez-Garcia et al., 2012b) and functional units have been considered. In particular, Carozzi et al. (2013a) evaluated the same solutions for soil tillage (conventional, minimum tillage and sod seeding) using a different method (fuzzy logic) based on environmental, production and cost variables. Although a direct comparison with this study cannot be done due to the different methodology adopted, it is interesting to underline that, also in this study, minimum tillage and sod seeding showed the best performance from an environmental point of view, while conventional

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

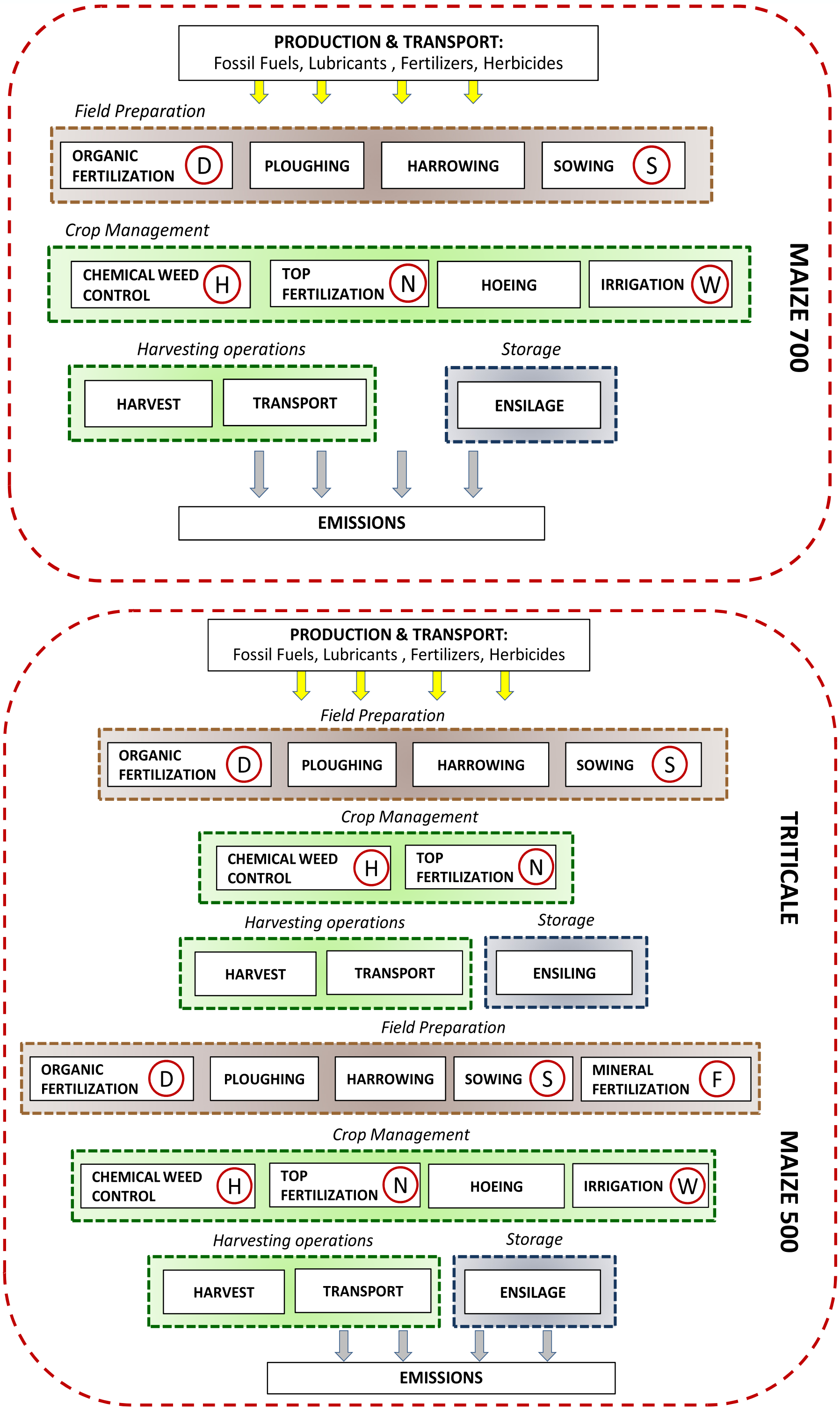


Fig. 2. System boundary of the analysis. On the top SC system (only Maize FAO 700), on the bottom DC system (triticale followed by maize FAO 500). The letters indicate the production factors used: D = digestate; S = seed; H = herbicide; N = nitrogen chemical fertilizer; W = irrigation water.

Table 3
Alternative scenarios regarding soil tillage and sowing.

Operations	Operative machine	Diesel fuel consumption (kg ha^{-1})		
		BS	MT	SS
Soil tillage	Plough	22.64	–	–
	Ripper	–	31.97	–
Harrowing	Rotary harrow	24.24	–	–
	Disk harrow	–	6.07	–
Sowing	Seed drill	2.68	2.68	–
No tillage	No till seed-drill	–	–	12.22

tillage achieved the best results when the maximization of the economic performances was aimed.

On the other hand, several LCA studies took into consideration different geographic areas (Kim et al., 2009; Wang et al., 2007; Munoz et al., 2014; Chong et al., 2014; Xialong et al., 2014):the crop cultivation practices are therefore different from the ones analysed in this study, leading to differences in the environmental impact. Nevertheless, when similar geographic areas are taken into account, the results of this study are in agreement

Table 4
SC and DC scores for all impact categories (FU = 1 t_{DM}).

Impact category		Unit	SC	DC
Abiotic depletion	AD	kg Sbeq	$3.52 \cdot 10^{-01}$	$4.48 \cdot 10^{-01}$
Acidification	AP	kg SO_2eq	$4.08 \cdot 10^{-00}$	$3.52 \cdot 10^{-00}$
Eutrophication	EP	kg PO_4eq	$1.07 \cdot 10^{-00}$	$8.49 \cdot 10^{-01}$
Climate Change	CC	kg CO_2eq	$8.89 \cdot 10^{1}$	$1.02 \cdot 10^{2}$
Ozone layer depletion	OD	kg CFC-11eq	$7.08 \cdot 10^{-06}$	$8.27 \cdot 10^{-06}$
Photochemical oxidation	POF	kg C_2H_4eq	$1.02 \cdot 10^{-02}$	$1.28 \cdot 10^{-02}$

with González-García et al. (2012b) with regard to the hotspots of the cereal cultivations. In more details both these studies highlighted that field emissions due to fertiliser applications are responsible for the main part of acidification and eutrophication.

As regard to the impact of 1 normal cubic meter of biomethane potential, the results of this study are similar to ones obtained by Bacenetti et al. (2014) who evaluated a DC system with wheat and maize FAO class 500; the differences are mainly due to the diverse biomass yields and different specific biomethane production values.

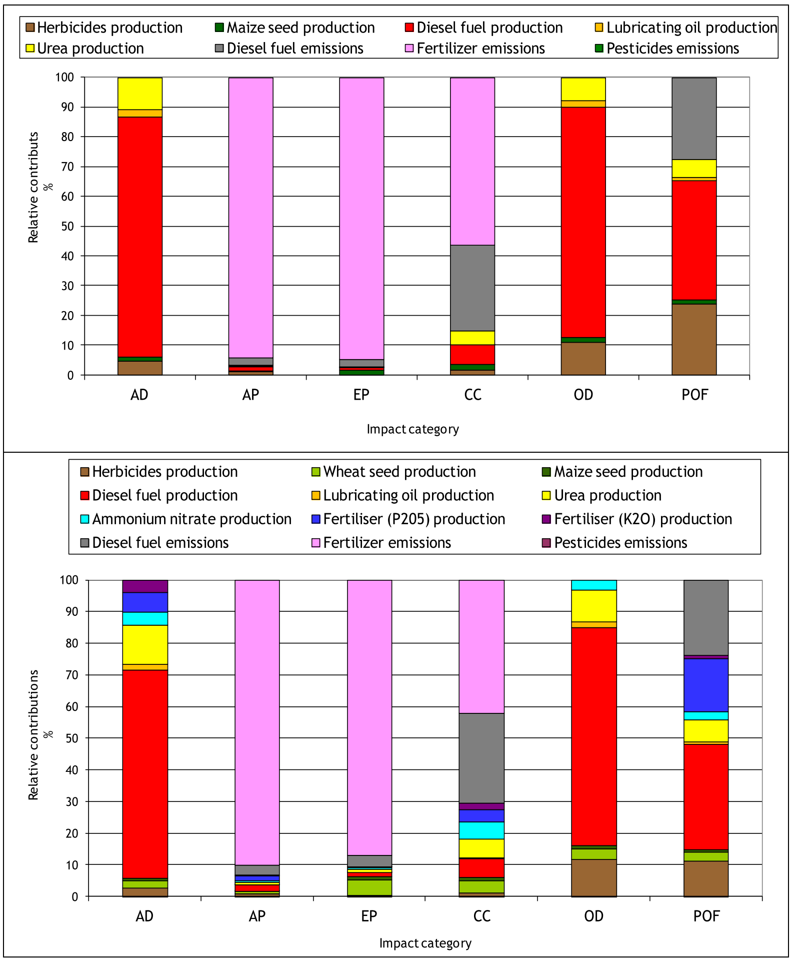


Fig. 3. Environmental burdens: impact of all inputs and emissions for the two cropping systems for the 6 considered impact categories. On the top SC system, on the bottom DC system.

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Table 5
Relative comparison among the different scenarios.

Impact category		Single crop (SC)			Double crop (DC)		
		BS	SS	MT	BS	SS	MT
Abiotic depletion	AD	100%	89.5%	97.5%	100%	88.9%	91.7%
Acidification	AP	100%	99.5%	99.8%	100%	99.3%	99.9%
Eutrophication	EP	100%	99.7%	99.9%	100%	99.4%	100.9%
Climate Change	CC	100%	96.1%	98.4%	100%	95.6%	101.1%
Ozone layer depletion	OD	100%	89.9%	97.6%	100%	88.3%	91.3%
Photochemical oxidation	POF	100%	92.6%	98.2%	100%	91.6%	96.8%

Another study conducted in Po Valley (Bacenetti et al., 2013a) showed, compared to our results, lower environmental impacts (except for acidification, for which similar values have been obtained) for single crop of maize, considering 1 ton of silage as functional unit. Such differences are due to biomass yield (lower in the present study) and nitrogen fertilisation (lower in the present analysis).

4.2. Alternative scenarios

Table 5 compares different scenarios of the two cropping systems (BS, MT, SS).

Generally, the simplification of field operations carried out for soil tillage slightly reduces the environmental burdens. This reduction is slightly higher for the SS scenario than for MT and, between the two cropping systems, for the DC (where soil tillage operations are carried out twice). In more details, for the scenario SS, greater environmental impact reductions are achieved for AD (−11% and −12% for SC and DC, respectively) and OD (−10% and −12% for SC and DC, respectively). The reduction is higher for these two impact categories because of their dependence to fossil fuel consumption. In the SC system, the environmental impacts of BS and MT are similar, while some relevant differences are observed in the SS scenario (for example −10.5% for AD with respect to BS). A similar trend can be identified for DC, except for EP and CC, for which MT shows slightly higher impacts than BS.

Fig. 4 shows the relative comparison between SC and DC and among the different scenarios considering the SC Baseline scenario as reference. Once again, it can be underlined that, for each cropping system, MT and SS allow a reduction of the environmental impact for all the impact categories under assessment. Nevertheless, the MT and SS scenarios for DC present higher impacts than the baseline scenario in SC for impact categories such as AD (+16.9% and 13.2% for MT and SS, respectively), CC (+16.4% and 10.0% for MT and SS, respectively), OD (+6.7% and 3.2% for MT and SS, respectively) and POF (+21.9% and 15.3% for MT and SS, respectively).

Regarding the assumption that zero change in soil carbon content, it should be underlined that, according to some studies (Smith et al., 1998; de Moraes Sà et al., 2001; Gao et al., 2007), conservative agriculture (SS and MT) can affect the soil carbon stocks and consequently the environmental impact (i.e. Climate Chance)of the cropping systems.

4.3. Selection of different functional unit

The choice of a particular FU significantly influences the environmental results (Nemecek et al., 2011). The function of the cropping systems under study is the production of silages for animal consumption as well as for energy production (biogas). Different functions and functional units can be used in agriculture, depending on the role played by the agricultural activity, including (i) the land management function, measured by cultivated hectare per year, (ii) the financial function, expressed per currency unit, and (iii) the productive function, quantified by physical units such as kg of dry matter or MJ net energy content. However, in this study, besides the dry matter production, other FUs have been selected for the comparison between SC and DC:

(1) 1 ha;
(2)1 normal cubic meter (1 m^3 at normal conditions) of potential methane (CH_4). The cereal silages are the most important feedstock used to feed the digesters of anaerobic digestion plants. The silages are characterized by different specific biomethane productions (Negri et al., 2014). Therefore, when these feedstock are used for energy purposes inside the anaerobic digesters, the biomethane production is a proper functional unit. The specific biogas productions were evaluated by means of laboratory tests following the methodology reported by Negri et al. (2014) and are equal to 102.3 m^3/t_{wb}, 105.6 m^3/t_{wb}, and 86.1 m^3/t_{wb} for silage of maize 700, maize 500, and triticale, respectively.

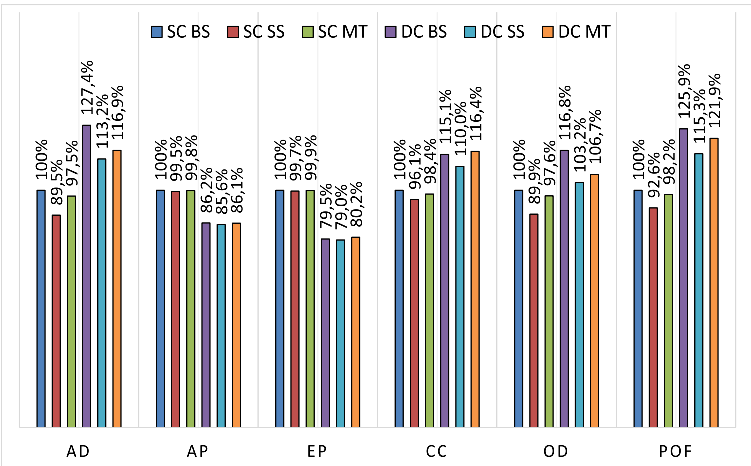


Fig. 4. Relative comparison among the different scenarios in the two cropping systems under evaluation (Single Crop Baseline Scenario = 100%).

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Table 6
Conversion to the different FU for 1 ha of cultivated area.

Crop system	Functional unit			
	Area ha	Dry matter yield t_{DM}	Biomethane m^3	Feed unit U_{FEED}
SC	1	25.41	7878	24,397
DC	1	33.17	8833	26,748

(3) 1 unit feed, a unit for measuring and comparing the nutritive value of feeds (established as the nutritive value of 1 kg of average quality dry barley) (McCartney and Vaage, 1994). The following feed units (U_{FEED}) were used in this study (accordingly with McCartney and Vaage, 1994; Maggiore, 2008): 960 U_{FEED}/t_{DM}, 940 U_{FEED}/t_{DM}, and 650 U_{FEED}/t_{DM} for silage of maize 700, maize 500, and triticale, respectively. Table 6 shows the conversion to the different FUs for 1 ha of the cultivated area.

Fig. 5 shows, for the different FUs, the comparison of the environmental results. It can be seen that, as predicted, in SC the scenarios MT and SS have a similar trend for all the selected FUs and, as already underlined, show environmental results comparable to BS.

Instead, for DC, generally, the environmental impact is higher than for SC. Moreover, for DC, changing the functional unit strongly affects the results/environmental performance. In particular, by using the mass of dry matter (1 t_{DM}) as FU, lower differences are relieved between the two cropping systems, while, oppositely, when the area (1 ha) is chosen as the FU, higher variations are found.

It is interesting to underline that for AP and EP, when the FU is the area, the DC shows higher environmental impact, while for the other 3 FUs, the SC system shows the highest values.

These differences are mainly due to the variation of the yield, specific biomethane production, and nutritive value and they highlight the importance of the FU choice. The selection of

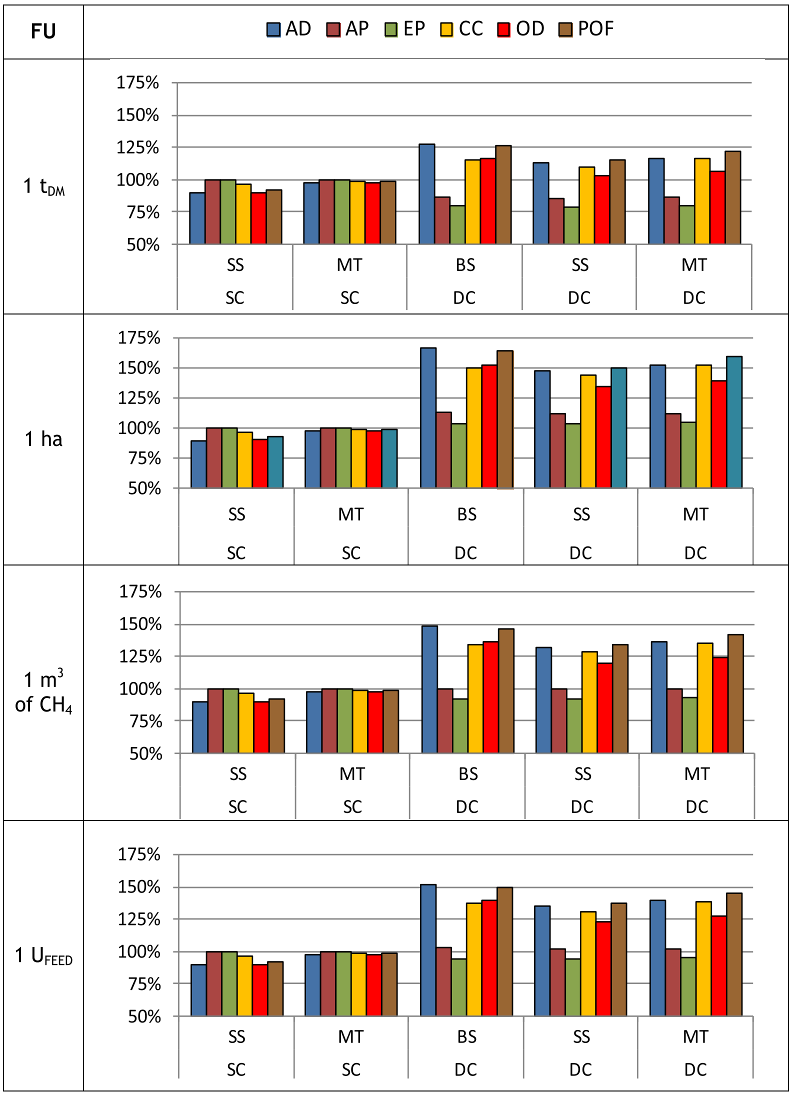


Fig. 5. Environmental impact for the different FU (Single Crop Baseline Scenario = 100%).

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

cultivated area as FU can be a proper choice if the land for farming is limited. Otherwise, selecting the area as FU is not a good solution to compare cropping systems which produce different amount of dry matter and which involve different cultivation practices. For the same reason the use the biomethane production or the unit feed as FU can better reflect the difference between cropping systems that, not only produce different amount of dry matter, but produce also biomass with different characteristics.

5. Conclusions

This study is focused on the evaluation of environmental burdens of the two most widespread cropping systems carried out in the Po Valley area for cereal silages production. In Italy, maize, among the summer cereals, and triticale, among the winter cereals, are the cereals most utilised for animal feeding and biogas production. This study assesses the environmental performances of the SC system (maize FAO class 700) and the DC system (triticale + maize FAO class 500), and it also considers different technical solutions for seed bed preparation.

The achieved results point out that for both cropping systems, the environmental burdens are mainly due to crop fertilisation (in particular, nitrogen application, primarily via organic fertilisers) and mechanisation of field operations (diesel fuel emissions and consumption). The emissions linked to fertiliser application are relevant especially for acidification and eutrophication. The organic fertilisation, carried out with large masses of digestate, involves high emissions of ammonia and nitrogen dioxide, especially because the application is performed on the soil's surface and without fast soil incorporation (Carozzi et al., 2013b).

Generally, the SC system shows better environmental performances for all the impact categories evaluated except for that of acidification and eutrophication. For these two impact categories, the DC system has lower environmental impact due to fewer fertiliser applications.

The comparison among the three different solutions for seedbed preparation highlights that, with respect to conventional soil tillage, minimum tillage and no tillage (sod seeding) can achieve better environmental performances. The reduction of the environmental load is mainly due to lower diesel fuel consumptions and it is higher in the DC system where seed bed preparation is carried out twice a year.

It should be noted that, the adoption of the above considered alternative solutions for soil tillage (minimum tillage and sod seeding) can affect the biomass yield in areas with particular pedoclimatc characteristics (e.g., clay and loam soil, strong presence of crop residues, etc.).

In order to perform a more precise and comprehensive assessment of crop system characterised by the different soil tillage solutions and crop residues management (Smith et al., 2012; Fusi et al., 2014) the effect of these practices on soil carbon stocks should be taken into consideration. For this purpose, specific field trials and measurements should be carried out.

Therefore, a future improvement will involve the analysis of the environmental impact of other cropping systems carried out in Po Valley considering also variation in soil carbon stocks.

Acknowledgements

Authors thank Regione Lombardia which financed a Postdoctoral Research Fellowship ("Progetto Dote Ricerca" financed by FSE – Regione Lombardia).

References

Audsley, E., 1997. Harmonization of Environmental Life Cycle Assessment for Agriculture. Final Report, Concerted Action AIR3-CT94-2028. European Commission, DG VI Agriculture, pp. 1–139.
Bacenetti, J., Fusi, A., Guidetti, R., Fiala, M., 2013a. Life Cycle Assessment of maize cultivation for biogas production. J. Agric. Eng. 44, 579–582.
Bacenetti, J., Fusi, A., Negri, M., Guidetti, R., Fiala, M., 2014. Environmental assessment of two different crop systems in terms of biomethane potential production. Sci. Total Environ. 466–467, 1066–1077.
Bacenetti, J., Mena, A., Negri, M., Cantarella, P., Bocchi, S., Fiala, M., 2012. Energetic and environmental balance of a biogas plant in northern Italy. In: Proceedings CIGR-AgEng 2012 International Conference on Agricultural Engineering Valencia: CIGR-AgEng.
Bacenetti, J., Negri, M., Fiala, M., Gonzalez Garcia, S., 2013b. Anaerobic digestion of different feedstock: impact on energetic and environmental balances of biogas process. Sci. Total Environ. 463–464, 541–551.
Basso, B., Sartori, L., Bertocco, M., Cammarano, D., Martin, E.,C., Grace, P., 2011. Economic and environmental evaluation of site-specific tillage in a maize crop in NE Italy. Eur. J. Agron. 35 (2), 83–92.
Bechini, L., Castoldi, N., 2009. On – farm monitoring of economic and environmental performances of cropping systems: results of a 2-year study at the field scale in northern Italy. Ecol. Indic. 9, 1096–1113.
Birkved, M., Hauschild, M., 2006. PestLCI—A model for estimating field emissions of pesticides in agricultural LCA. Ecol. Model 198, 433–451.
Borrelli, L., Castelli, F., Ceotto, E., Cabassi, G., Tomasoni, C., 2014. Maize grain and silage yield and yield stability in a long-term cropping system experiment in Northern Italy. Eur. J. Agron. 55, 12–19.
Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an Input to LCA studies in the agricultural sector. Int. J. Life Cycle Assess. 5, 349–357.
Brentrup, F., Küsters, J., Kuhlmann, H., Lammel, J., 2001. Application of the Life Cycle Assessment methodology to agricultural production: an example of sugar beet production with different forms of nitrogen fertilisers. Eur. J. Agron. 14 (3), 221–233.
Capponi, S., Fazio, S., Barbanti, L., 2012. CO2 savings affect the break-even distance of feedstock supply and digestate placement in biogas production. Renew. Energy 37, 45–52.
Carozzi, M., Bregaglio, S., Scaglia, B., Bernardoni, E., Acutis, M., Confalonieri, R., 2013a. The development of a methodology using fuzzy logic to assess the performance of cropping systems based on a case study of maize in the Po Valley. Soil. Use Manag. 29, 576–585.
Carozzi, M., Ferrara, R.M., Rana, G., Acutis, M., 2013b. Evaluation of mitigation strategies to reduce ammonia losses from slurry fertilization on arable lands. Sci. Total Environ. 449, 126–133.
Carrosio, G., 2013. Energy production from biogas in the Italian countryside: policies and organizational models. Energy Policy 63, 3–9.
Castanheira, E.,G., Freire, F., 2013. Greenhouse gas assessment of soybean production: implications of land use change and different cultivation systems. J. Clean. Prod. 54, 49–60.
Cherubini, F., Bargigli, S., Ulgiati, S., 2009. Life cycle assessment (LCA) of waste management strategies: landfilling, sorting plant and incineration. Energy 34, 2116–2123.
Chong, W., XiaoLin, Li, TingTing, G., et al., 2014. Life cycle assessment of wheat-maize rotation system emphasizing high crop yield and high resource use efficiency in Quzhou County. J. Clean. Prod. 68, 56–63.
de Moraes Sà, J.C.M., Cerri, C.C., Dick, W.A., Lal, R., Filho, S.P.V., Piccolo, M.C., Feigl, B.E., 2001. Organic matter dynamics and carbon sequestration rates for a tillage chronosequence in a Brazilian Oxisol. Soil. Sci. Soc. A. J. 65, 1486–1499.
DETEC (Federal Department of the Environment, Transport, Energy and Communications), http://www.bafu.admin.ch/index.html?lang=it last visit 05/04/2014.
Deytieux, V., Nemecek, T., Knuchel, R.F., Gaillard, G., Munier-Jolain, N.M., 2012. Is integrated weed management efficient for reducing environmental impacts of cropping systems? A case study based on life cycle assessment. Eur. J. Agron. 36 (1), 55–65.
Fiala, M., Bacenetti, J., 2012. Model for the economic, energetic and environmental evaluation in biomass productions. J. Agr. Eng. 42, 26–35.
Food and Agriculture Organization of the United Nations (FAO), 2013. Food Outlook – Biannual Report on Global Food Markets. ISSN: 0251-1959, pp. 1–134 (print).
Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G., 2007. Overview and Methodology. Ecoinvent report No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf.
Fusi, A., Bacenetti, J., González-García, S., Vercesi, A., Bocchi, S., Fiala, M., 2014. Environmental profile of paddy rice cultivation with different straw management. Sci. Total Environ. 494–495, 119–128.
Gao, H.W., Wu, H.D., Li, W.Y., Wang, X.Y., He, J., 2007. Effects of 15 years of conservation tillage on soil structure and productivity of wheat cultivation in northern China. Aust. J. Soil. Res. 45, 344–350.
Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A., Tempio, G., 2013. Tackling Climate Change through Livestock – a Global Assessment of Emissions and Mitigation Opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome.

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Giunta, F., Motzo, R., 2004. Sowing rate and cultivar affect total biomass and grain yield of spring triticale (×Triticosecale Wittmack) grown in a Mediterranean-type environment. Field Crops Res. 87 (2–3), 179–193.
Goedkoop, M., de Schryver, A., Oele, M., 2010. Introduction to LCA with SimaPro 7. PRé Consultants, the Netherlands.
González-García, S., Bacenetti, J., Murphy, R.J., Fiala, M., 2012a. Present and future environmental impact of poplar cultivation in Po Valley (Italy) under different crop management systems. J. Clean. Prod. 26, 56–66.
González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L., 2012b. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J. Clean. Prod. 43, 71–83.
Guinée, J.B. (Ed.), 2002. Handbook on Life Cycle Assessment, Operational Guide to the ISO Standards, pp. 1–708.
IPCC, 2006. Agriculture, forestry and other land use. In: Eggleston, et al. (Eds.), IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme.
Iriarte, A., Rieradevall, J., Gabarrell, X., 2010. Life cycle assessment of sunflower and rapeseed as energy crops under Chilean conditions. J. Clean. Prod. 18 (4), 336–345.
ISO, 2006. Environmental Management-Life Cycle Assessment-Principles and Framework (ISO 14040). European Committee for Standardization, Brussels.
ISTAT – Istituto Nazionale di Statistica, 2010. Sixth General Census of Agriculture. Available at: http://www.istat.it/it/censimento-agricoltura/agricoltura-2010.
ISTAT – Istituto Nazionale di Statistica, 2012. Tavola C02-Superficie (ettari) e produzione (quintali): riso, mais, sorgo, altri cereali. Dettaglio per regione.
Kennedy, T.F., Connery, J., Fortune, T., Forristal, D., Frant, J., 2013. A comparison of the effects of minimum-till and conventional-till methods, with and without straw incorporation, on slugs, slug damage, earthworms and carabid beetles in autumn-sown cereals. J. Agric. Sci. 151 (5), 605–629.
Kim, S., Dale, B.E., Jenkins, R., 2009. Life cycle assessment of corn grain and corn stover in the United States. Int. J. Life Cycle Assess. 14, 160–174.
Kustermann, B., Munch, J.C., Hulsbergen, K.J., 2013. Effects of soil tillage and fertilization on resource efficiency and greenhouse gas emissions in a long-term field experiment in Southern Germany. Eur. J. Agron. 46, 61–73.
Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Lema, J.M., Moreira, M.T., 2014a. Life Cycle Assessment of electricity production in Italy form anaerobic co-digestion of pig slurry and energy crops. Renew. Energ. 68, 625–635.
Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Moreira, M.T., 2014b. Assuring the sustainable production of biogás from anaerobic mono-digestion. J. Clean. Prod. 72, 23–34.
Maggiore, T., 2008. Attività di valutazione di mais per destinazione energetica. Regione Lombardia, pp. 1–64. www.ersaf.lombardia.it/upload/ersaf/pubblicazioni/Relazione%20I.II%20Anno_02_13383_692.pdf.
Mangena, S.J., Brent, A.C., 2006. Application of a Life Cycle Impact Assessment framework to evaluate and compare environmental performances with economic values of supplied coal products. J. Clean. Prod. 14 (12–13), 1071–1084.
McCartney, D.H., Vaage, A.S., 1994. Comparative yield and feeding value of barley, oat and triticale silages. Can. J. Animal Sci. 74 (1), 91–96.
Milà i Canals, L., Burnip, G.M., Cowell, S.J., 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): case study in New Zealand. Agric. Ecosyst. Environ. 114 (2 – 4), 226–238.
Munoz, I., Flury, K., Jungbluth, N., Rigarlsford, G., Mila i Canals, L., King, H., 2014. Life cycle assessment of bio-based ethanol produced from different agricultural feedstocks. Int. J. Life Cyc. Assess. 19 (1), 109–119.
Negri, M., Bacenetti, J., Brambilla, M., Manfredini, A., Cantore, C., Bocchi, S., 2014a. Biomethane production from different crop systems of cereals in Northern Italy. Biomass Bioener 63, 321–329.
Negri, M., Bacenetti, J., Manfredini, A., Lovarelli, D., Maggiore, T., Fiala, M., Bocchi, S., 2014b. Evaluation of methane production from maize silage by harvest of different plant portions. Biomass Bioener. 67, 339–346.
Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104, 217–232.
Poeschl, M., Ward, S., Owende, P., 2012a. Environmental impacts of biogas deployment – Part I: life cycle inventory for evaluation of production process emissions to air. J. Clean. Prod. 24, 168–183.
Poeschl, M., Ward, S., Owende, P., 2012b. Environmental impacts of biogas deployment – Part II: life cycle assessment of multiple production and utilization pathways. J. Clean. Prod. 24, 184–201.
Santilocchi, R., Bianchelli, M., 2006. Mais su sodo nelle Marche funziona. L'informatore Agrar. 16, 51–53.
Santin-Montanya, M.I., Martin-Lammerding, D., Walter, I., Zambrana, E., Tenorio, J.L., 2013. Effects of tillage, crop systems and fertilization on weed abundance and diversity in 4-year dry land winter wheat. Eur. J. Agron. 48, 43–49.
Smil, V., 2000. Phosphorus in the environment: natural flows and human interferences. Annu. Rev. Energ. Env. 25, 53–88.
Smith, P., Powlson, D.S., Glendenning, M.J., Smith, J.U., 1998. Preliminary estimates of the potential for carbon mitigation in European soils through no-till farming. Glob. Change Biol. 4, 679–685.
Smith, W.N., Grant, B.B., Campbell, C.A., McConkey, B.G., Desjardins, R.L., Krobel, R., Malhi, S.S., 2012. Crop residue removal effects on soil carbon: measured and inter-model comparisons. Agric. Ecosyst. Environ. 161, 27–38.
Uchida, S., Hayashi, K., 2012. Comparative life cycle assessment of improved and conventional cultivation practices for energy crops in Japan. Biomass Bioenerg. 36, 302–315.
Van der Werf, H.M.G., Tzilivakis, J., Lewis, K., Basset-Mens, C., 2007. Environmental impacts of farm scenarios according to five assessment methods. Agric. Ecosyst. Environ. 118, 327–338.
Wang, M.X., Wu, W.L., Liu, W.N., Bao, Y.H., 2007. Life cycle assessment of the winter wheat-summer maize production system on the North China Plain. Int. J. Sust. World 14, 400–407.
XiaoLong, W., YuanQuan, C., Sui, Peng, 2014. Emergy analysis of grain production systems on large-scale farms in the North China Plain based on LCA. Agric. Syst. 128, 66–78.
Zhang, M.Y., Wang, F.J., Chen, F., Malemela, M., Zhang, H.L., 2013. Comparison of three tillage systems in the wheat-maize system on carbon sequestration in the North China Plain. J. Clean. Prod. 54, 101–107.

Please cite this article in press as: Bacenetti, J., et al., Impact of cropping system and soil tillage on environmental performance of cereal silage productions, Journal of Cleaner Production (2014), http://dx.doi.org/10.1016/j.jclepro.2014.08.052

Evaluation of environmental impacts in the catering sector: The case of pasta

Alessandra Fusi[1,2], Riccardo Guidetti[1] and Adisa Azapagic[2*]

[1]Department of Agricultural and Environmental Sciences - Production, Landscape, Agroenergy, Università degli Studi di Milano, Via Celoria 2, 20133, Milano, Italy
[2]School of Chemical Engineering and Analytical Science, The Mill, Sackville Street, The University of Manchester, Manchester, M13 9PL, UK

*Corresponding author:
Email: adisa.azapagic@manchester.ac.uk
Tel: +44 (0) 161 306 4363

Abstract

Despite its enormous size and economic value, there is currently scant information on environmental impacts from the catering sector. At the same time, the awareness of and preferences for environmentally sustainable food preparation and consumption are growing. In general, two catering approaches are practiced: cook-serve and deferred. In the former, food is cooked and immediately served to consumers while the latter allows for the food to be prepared at times and places completely different from consumption. This study focuses on environmental impacts of deferred catering with the aim of evaluating different options for food preparation and distribution, to help identify environmentally sustainable solutions. For these purposes, the case of pasta, one of the most popular foods worldwide, is considered. Two main types of deferred system (cook-warm and cook-chill) and cooking technologies (pasta cookers and range tops) used in the catering sector are evaluated. The results suggest that cooking in pasta cookers saves up to 60% of energy and 38% of water compared to range tops and therefore reduces by 34-66% the impacts associated with pasta preparation. The environmental impacts of pasta cooking could also be reduced by using gas rather than electric appliances as the impacts of the latter are higher by 13-98%. Pasta cooking is the major hotspot in both the cook-chill and cook-warm chains. Overall, the impacts from the cook-chill chain are 17-96% higher than from the cook-warm system, mainly because of the use of refrigerants and higher consumption of energy.

Keywords: catering sector; cook-warm chain; cook-chill chain; food preparation; environmental impacts; life cycle assessment

1. Introduction

Catering is a complex system involving both people and equipment in the preparation and serving of food. Such systems transform a diverse combination of inputs into desired outputs (Smith and West, 2003). A commonly accepted definition of the term “food service” or “catering” is “the provision of food and beverages away from home*”* (Davis *et al.* 1998). Traditionally, catering has been divided into the “cost food service sector” or “contract catering”, which, broadly speaking, refers to not-for-profit catering activities, and the “profit sector” (Smith and West, 2003). The former includes catering outlets for business, education and health care, while the latter comprises profit-orientated establishments as restaurants, fast-food chain outlets, cafes, takeaways, pubs, leisure and travel catering outlets (Bourlakis and Weightman, 2004).

In general, two catering approaches are practiced: conventional or cook-serve and deferred (Ciappellano, 2009). In the former, food is cooked and immediately served to consumers with all stages of food preparation occurring in a few hours before the food is served and consumed. This is typically the case in restaurants and canteens. The deferred system, on the other hand, allows for the food to be prepared at times and in places completely separated from consumption: here, the food preparation and cooking are carried out in centralised kitchens, from which the prepared meals are distributed to consumers (e.g. hospitals, schools, companies, etc.). The time difference between the preparation in the catering centre and the consumption can be several hours, days or even months, depending on the method used to preserve the food. Two main types of deferred system can be distinguished: the cook-warm and cook-chill chains (Williams,1996; Ciappellano, 2009; Risteco, 2006a). In cook-warm chains, the food is distributed at a temperature of 65°C (to

1

avoid the risk of microbial growth) and the consumption should occur within two hours after cooking (Ciappellano, 2009; Epicentro, 2012). The cook-chill system is defined as “a catering system based on the full cooking of food followed by fast chilling and storage in controlled low-temperature conditions above the freezing point, usually 0-3°C” (Evans et al., 1996). The aim of the cooking process in the cook-chill system is to ensure destruction of vegetative stages of any pathogenic micro-organisms (Evans et al., 1996).

The contract catering sector in Europe employs over 600,000 people and delivers over 6 billion meals each year (Ferco, 2014). This equates to 67 million consumers served every day, or one in four meals eaten outside the home (Ferco, 2014). In Italy alone, the contract catering sector is worth €6.2 billion to which the health care sector (hospitals, nursing homes) contributes 34%, the education sector 30% and catering for business the remaining 36% (Angem, 2014).

Yet, despite its enormous size and economic value, there is currently scant information on environmental impacts of the catering sector. At the same time, the awareness of and preferences for environmentally sustainable practices for food preparation and consumption are growing. This is largely driven by the need to reduce costs but also to gain market advantage by attracting environmentally conscious consumers (Baldwin et al., 2011). Therefore, in an attempt to contribute towards a better understanding of environmental impacts in the catering sector, this study focuses on the deferred system with the aim of evaluating different options for food preparation and distribution, to help identify environmentally sustainable solutions. As an example, the study considers pasta, one of the most popular foods worldwide. Both the cook-chill and cook-warm chains are included in the analysis. While the findings are specific to the pasta, they could be applicable to some other foods as the technologies and approaches used in the catering sector are similar. The outcomes of such analysis could be helpful to food-service providers in planning more sustainable catering activities as well as to consumers in choosing more sustainable providers.

2. Methodology

Life cycle assessment (LCA) has been used to estimate the environmental impacts of pasta cooking and distribution to consumers, following the ISO 14040/44 methodology (ISO, 2006a; b). The goal of the study and the data used are detailed in the sections below, together with the assumptions.

2.1 Goal and scope of the study

The aim of this study is twofold:

i) to evaluate the environmental impacts associated with the preparation (cooking) of pasta in professional kitchens and compare different cooking technologies used most-commonly in the catering sector; and
ii) to compare the impacts of the cook-warm and the cook-chill chains in the deferred catering system.

The following cooking technologies are considered:

- pasta cookers: electric, gas and liquefied petroleum gas (LPG); and
- range tops (stove hobs): gas, electric, infrared and induction.

The stages typically involved in the cook-warm and cook-chill chains are outlined in Figure 1 and the system boundaries considered in the study are given in Figure 2. As can be seen from the latter, the following activities are included in the study:

- pasta cooking;
- for the cook-chill chain: blast chilling, refrigerated storage, refrigerated transportation to the consumer and regeneration (reheating of pasta); and
- for the cook-warm chain: ambient transport to the consumer.

The environmental impacts of the production of pasta, which is common to all cooking methods and chain management approaches, are excluded from the study. Similarly, the packaging, food serving and post-consumer waste management are not considered as they are present in both the cook-warm and cook-chill chains. Furthermore, the emissions of particulate matter and sulphur dioxide generated during pasta cooking are also excluded as they are very low (Zhang et al., 2010; EPA, 2014). The impacts of the manufacture of pasta cookers and range tops are not considered as their contribution over the life time would be negligible.

The functional unit is defined as the "preparation and distribution of 1 kg of cooked pasta". Spaghetti are considered as an example but a similar catering approach and findings would apply to other types of pasta. To obtain 1 kg of cooked pasta, 444 g of dry pasta is needed. The study is based in Italy.

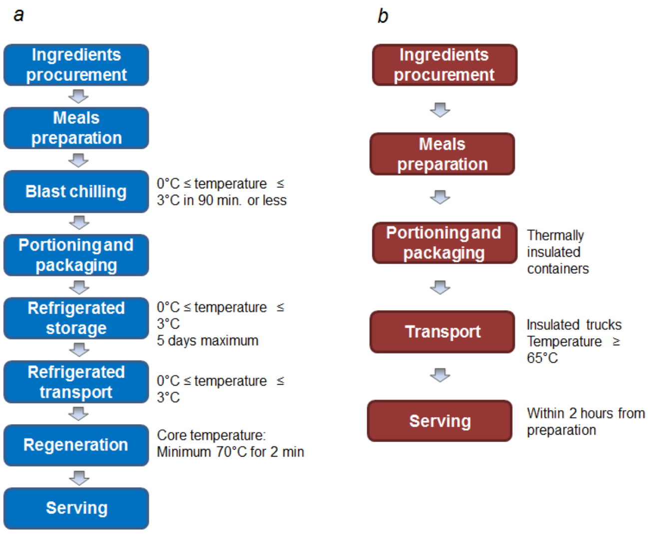


Figure 1. Activities involved in the cook-chill (a) and cook-warm chains (b).

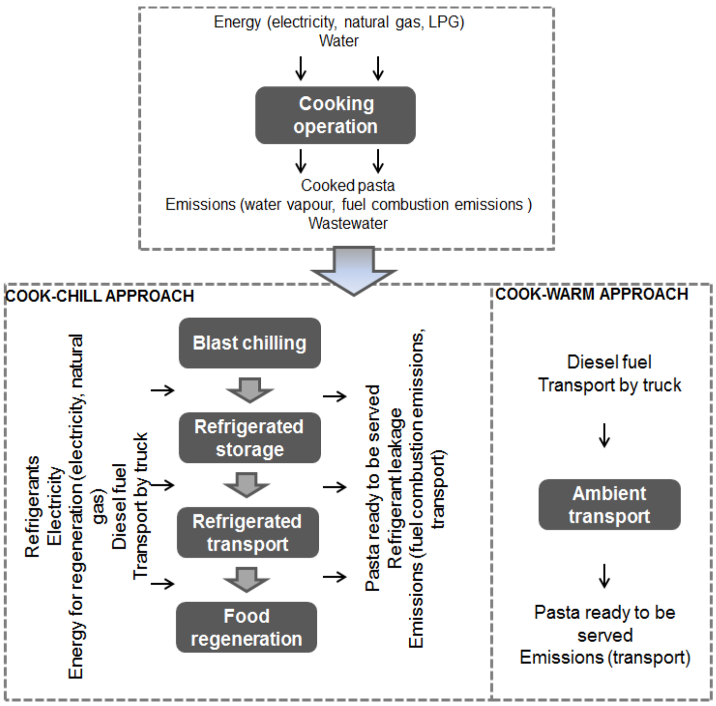


Figure 2. System boundaries considered in the study.

3

2.2 Inventory data

This section specifies the assumptions and data used in different life cycle stages, starting with the cooking and followed by cook-chill and cook-warm chains, respectively.

2.2.1 Cooking stage

The inventory data for cooking the pasta in cookers and range tops are summarised in Table 1 and Table 2, respectively. As can be seen, the following inputs and outputs are considered:

- water to cook the pasta;
- energy required to heat the tap water to 100°C;
- energy required to cook dry pasta for 8 minutes (time based on pasta producers' specification and the study carried out by Marti et al., 2013);
- water vapour produced during cooking;
- wastewater disposed of after the cooking; and
- CO_2, CH_4, CO and NOx emissions from natural gas and (LPG) pasta cookers and range tops.

The data have been obtained from various sources, including scientific literature, manufacturers' specifications, legislation and personal communication with a cooking centre. Background data have been sourced from Ecoinvent v. 2.2 (Frischknecht et al., 2007) and ILCD (Wolf et al., 2012).

Table 1 Inventory data for cooking in pasta cookers.

	Unit[a]	Electric	Gas	LPG	Data sources
Inputs					
Water[b]	kg	4.44	4.44	4.44	Manufacturers specification[c]
Heating energy[d]	MJ	1.018	1.969	1.969	FSTC (1999), CEN (2005)[c], Manufacturer specification[c]
Cooking energy[e]	MJ	0.519	0.823	0.823	Marti et al. (2013)[c], Manufacturers specification[c]
Outputs					
Water vapour	kg	0.22	0.18	0.18	See Appendix A for details
Wastewater	kg	1.99	2.01	2.01	Marti et al. (2013)[c], FSTC (1999)[c]
BOD	g	0.597	0.603	0.603	Presidente della Repubblica (2011)
CO_2	g	-	156.62[f] (151.61-162.78)	176.15[f] (172-183.16)	IPCC (2006)
CH_4	mg	-	2.80[f] (0.84-8.4)	2.80[f] (0.84-8.4)	IPCC (2006)
CO	g	-	0.08[f] (0.05-0.12)	0.14[f] (0.08-0.19)	EMEP/EEA (2013)
N_2O	mg	-	0.28[f] (0.08-0.83)	0.28[f] (0.08-0.83)	EMEP/EEA (2013)
NOx	g	-	0.14[f] (0.07-0.56)	0.14[f] (0.08-0.19)	EMEP/EEA (2013)

[a]All units per 1 kg of cooked pasta.
[b]The mass of water is related to the mass of dry pasta which is cooked at a ratio of water:pasta=10:1 (Marti et al., 2013; Ruini et al, 2013a).
[c]Data shown in the table calculated based on the original data from these sources. For data from manufacturers see Figure 3.
[d]Energy needed to bring water to boil from tap water temperature of 14.5ºC (the latter sourced from Metropolitana Milanese SPA, 2014).
[e]Energy required to cook dry pasta for 8 minutes.
[f] Default value reported in the respective references with the minimum and maximum values shown in brackets.

Table 2 Inventory data for pasta cooking on range tops.

	Unit[a]	Power rating available on the market as specified in Errore. L'origine riferimento non è stata trovata. Minimum Electric	Infrared	Induction	Gas	Maximum Induction	Gas	Data sources
Inputs								
Water[b]	kg	4.44	4.44	4.44	4.44	4.44	4.44	FSTC (2002)[c]
Heating[d] energy	MJ	2.893	2.743	1.768	3.060	1.768	3.060	CEN, 2005[c]; Manufacturers specification[c]
Cooking[e] energy	MJ	0.705	0.705	0.705	0.705	1.176	1.622	Marti et al. (2013)[c]; Manufacturers specification[c]
Outputs								
Water vapour	kg	0.17	0.18	0.28	0.16	0.47	0.44	
Wastewater	kg	3.72	3.71	3.61	3.73	3.42	3.52	Marti et al. (2013)[c]
BOD	g	1.12	1.11	1.08	1.12	1.03	1.06	Presidente della Repubblica (2011)
CO_2	g	-	-	-	210.47[f] (204.4-219.5)	-	262.61[f] (254.2-272.9)	IPCC (2006)
CH_4	mg	-	-	-	3.92[f] (1.13-11.3)	-	4.73[f] (1.4-14)	IPCC (2006)
N_2O	mg	-	-	-	0.38[f] (0.11-1.13)	-	0.47[f] (0.14-1.4)	EMEP/EEA (2013)
NOx	g				0.19[f] (0.09-0.75)		0.23[f] (0.12-0.94)	EMEP/EEA (2013)
CO	g				0.11[f] (0.07-0.16)		0.14[f] (0.08-0.2)	EMEP/EEA (2013)

[a]All units per 1 kg of cooked pasta.
[b]The mass of water is related to the mass of dry pasta which can be cooked at a ratio of water:pasta=10:1 (Marti et al., 2013; Ruini et al, 2013a).
[c]Data shown in the table calculated based on the original data from these sources. For data from manufacturers, see Table 3.
[d]Energy needed to bring water to boil from tap temperature of 14.5ºC (the latter sourced from Metropolitana Milanese SPA, 2014).
[e]Energy required to cook dry pasta for 8 minutes.
[f] Default value reported in the respective references with the minimum and maximum values shown in brackets.

Table 3 Power rating of range tops assumed in the study.

Type of range tops[a]	Power rating according to manufacturers (kW)	Classification of range tops by FSTC (2002) according to power rating (kW)
Electric (min-max)	1.5-4[b]	< 4.7
Electric infrared (min-max)	2.1-3.4[c]	< 4.7
Electric induction (min)	3.5	< 4.7
Electric induction (max)	5	>4.7 & < 7.62
Gas (min)	1.5	< 4.7
Gas (min)	10	> 7.62

[a] 'Min' and 'max' refers to the minimum and maximum burner size available on the market
[b] 1.5 kW used in the calculations.
[c] 2.1 kW used in the calculations.

The energy needed to boil the water (E_{heat}) and to cook the pasta (E_{cook}) given in Table 1 and Table 2 have been calculated according to equations (1) and (2), respectively:

$$E_{heat} = \frac{q \ x \ \Delta T \ x \ c_p}{\eta \ x \ C} \quad (kJ/kg) \qquad (1)$$

$$E_{cook} = \frac{P \ x \ t}{C} \quad (kJ/kg) \qquad (2)$$

where:

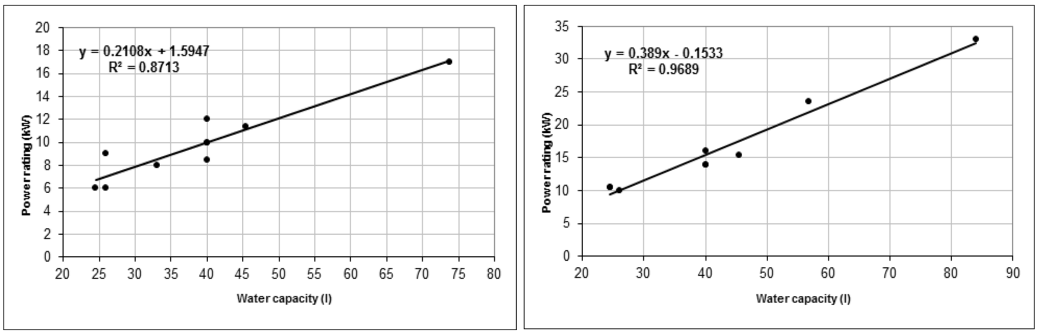
q	the amount of water needed to cook the pasta (l)
ΔT	the difference between the initial temperature of the water (14.5ºC) and the boiling temperature (100°C)
c_p	specific heat of water (4.186 kJ/kg °C)
η	cooking efficiency of the appliances, defined as the ratio of the energy transferred to the water to the energy consumed by the appliance
P	power rating of pasta cooker or range top (kW)
t	time to cook pasta (8 mins or 480 s).
C	capacity of pasta cooker or a pot used on range tops.

The above variables have been obtained as follows:

- Power rating (P) and capacity (C) for pasta cookers: a range of data have been collected from manufacturers of commercial pasta cookers dominating the catering market (for data points, see Figure 3).
- Power rating (P) for range tops: using manufacturers' data, the power rating has been identified for each type of range tops dominating the market (see Table 3). These values have been classified according to the different burner size shown in Table 3. In cases where for the same burner size a range of power ratings were found (electric and infrared range tops), the energy needed to cook pasta has been estimated assuming the minimum value for that burner size category as the energy they provide is sufficient for the specified amount of water (and pasta); in any case, if using burners with a higher power rating, the power input can be reduced to the minimum needed for cooking to save energy.
- Cooking efficiency (η): data on the efficiencies of the appliances have been obtained from literature (see Table 4).
- The amount of water needed to cook the pasta (q): a relationship between the power rating of the appliances and the associated amount of water has been defined as follows. For pasta cookers, an equation describing the relationship between power rating and water capacity has been defined using manufacturers' specification (see Figure 3). In the base case, the mean capacity has been assumed; the influence of different cooker sizes on the environmental impacts of cooking is explored through a sensitivity analysis later in the paper. For the range tops, the amount of water to cook the pasta is related to the capacity of pasta pots (C) and has been determined according to the size of the burners as shown in Table 5.

Furthermore, a two-cycle cooking process has been assumed for pasta cookers [1]. This means that part of the water used to cook pasta in the first cooking cycle is re-used to cook another batch of pasta, with the addition of fresh water to compensate for water losses through evaporation and absorption by pasta. The energy required to heat the water to the boiling point is lower for the second cycle, since the temperature of the water in the cooker is higher than in the first cycle (see Appendix A for estimates). Thus, the water use, heating energy and wastewater have been averaged over the two cycles and these data have been used for LCA modelling (see Table 1 and Appendix A).

[1] Personal communication with an Italian cooking centre.



a) b)

Figure 3 The relationship between water capacity and power rating for electric (a) and gas (b) pasta cookers estimated using manufacturers' data.
[Note that some of the points overlap so that there are more data than visible in the graphs.]

Table 4 Efficiencies of cooking appliances.

	Cooking efficiency (%)[a]	Source
Pasta cooker - electric	97.4	Average from FSTC (1999)
Pasta cooker - gas/LPG	50	CEN (2005)[b]
Range top – electric	55	Museo Energia (2013); Manufacturer (2014)[c]
Range top - infrared	58	Museo Energia (2013); Manufacturer (2014)[c]
Range top - induction	90	Museo Energia (2013); Manufacturer (2014)[c]
Range top – gas	52	CEN (2008)[b]

[a]Cooking efficiency is assumed to be constant for all sizes of pasta cookers and top ranges.
[b]Minimum requirement.
[c]Confidential.

Table 5 Water capacity of pasta pots according to the classification of range tops by FSTC (2002) based on the power rating.

Power rating of range tops (kW)	Water capacity of pasta pot (l)
< 4.7	4.54
>4.7 and < 7.62	9.07
> 7.62	13.15

2.2.2 Cook-chill chain

As shown in Figure 1, following pasta cooking, the cook-chill chain involves blast chilling, refrigerated storage, refrigerated transportation and regeneration (or reheating of pasta). The data and assumptions for these stages are described in the following sections.

2.2.2.1 Blast chilling

Following a similar approach as for the cooking appliances described in the previous section, technical data of a representative sample of commercial blast chillers dominating the market have been collected to define a relationship between the capacity and power rating for these appliances (see Figure 4). Like the cooking energy, the energy requirement for blast chilling has been estimated based on the power rating and capacity of blast chillers in Figure 4 and the time of 1 hour[1] needed to cool the pasta to 3°C:

$$E_{chill} = \frac{P_{chill}\ x\ t_{chill}}{C_{chill}} \qquad (kJ/kg) \qquad (3)$$

where:
E_{chill} energy required to cool 1 kg of cooked pasta to 3°C
P_{chill} power rating of the blast chiller (Figure 4) (kW)
t_{chill} time required to cool the pasta (1h)
C_{chill} capacity of the blast chiller (kg).

These results are shown in Table 6 for the mean capacity of the chiller, with a sensitivity analysis exploring later in the paper the influence of different chiller sizes on the environmental impact from this stage. Note that the estimated energy for blast chilling of 50 kWh/t of product (Table 6) agrees well with the range of 70-130 kWh/t of product reported by Duiven and Binard (2002) for blast freezing, taking into account that the energy consumption is higher for the latter than the former.

The refrigerant used in blast chillers is assumed to be R404A and the LCA data for its manufacture are based on the study by Bovea et al. (2007). The expected leakage of refrigerant is 5-10% per year[2] so that an average value of 7.5% has been assumed. The amount of refrigerant leaked during the time needed to chill the pasta (1 hr) has been estimated assuming that the blast chiller is switched on for 8 hours per day**Errore. Il segnalibro non è definito.** over 254 working days per year, as shown Table 6.

2.2.2.2 Refrigerated storage and transport

Cooked pasta can be stored in refrigerators from one to five days. Two types of refrigerant have been considered for the refrigerated storage – R404A and ammonia – assuming an annual leakage of 15% (DEFRA, 2008). The energy consumption during the storage is assumed at 0.26 Wh/kg·h (DEFRA, 2008). Table 7 shows the estimates for these parameters for different storage time.

The data for refrigerated transport are summarised in Table 8. In the base case analysis, the chilled pasta is assumed to be transported to the consumer by a 20-28 t truck over an average distance of 50 km; shorter (1km) and longer (100 km) distances as well as different vehicle sizes are considered within a sensitivity analysis. The life cycle inventory data for transport have been sourced from Ecoinvent (Frischknecht et al., 2007) but have been modified to include the additional amount of fuel (and the emissions) used by the refrigeration unit as well as the production and leakage of refrigerants, with the latter assumed at 22.5% of the annual charge (DEFRA, 2008; UNEP, 2003). The LCA data for the production of different types of refrigerant (R404A, R134A, R410A) have been sourced from Bovea et al. (2007).

2.2.2.3 Regeneration (reheating)

The following appliances have been considered for reheating: gas and electric combination oven, which are the most-widely used appliances in professional kitchens (Rohatsch et al., 2007) and microwave ovens, which are increasingly used in establishments where fast heating is required as well as in the hospitality industry (Rohatsch et al., 2007). The energy consumption for reheating shown in Table 9 has been estimated based on the oven pre-heating requirements, equal to 15% of the total energy needed for reheating in combination ovens (FSTC, 2002), the heating time of 7 minutes for combination and 65 seconds for microwave ovens (Rohatsch et al., 2007) and the temperature of 70°C that must be reached to avoid bacterial contamination (Ciappellano, 2009). The CO_2 and CH_4 emissions associated with gas combustion have been calculated using the IPCC emission factors (IPCC, 2006) while N_2O, CO and NOx emissions have been estimated according to EMEP/EEA (2013).

Table 6 Inventory data for blast chilling.

	Amount
Energy consumption (Wh/kg$_{\text{cooked pasta}}$)	50[a]
Refrigerant load (mg/kg$_{\text{cooked pasta}}$)	60[b]
Refrigerant leakage (mg/kg$_{\text{cooked pasta}}$)	4.5[b]

[a]Estimated using eq. (3) and the relationship in Figure 4, assuming the mean power rating for blast chillers.
[b] Source: personal communication with Professor Savvas Tassou, Brunel University.

[2] Personal communication with Professor Savvas Tassou, Brunel University.

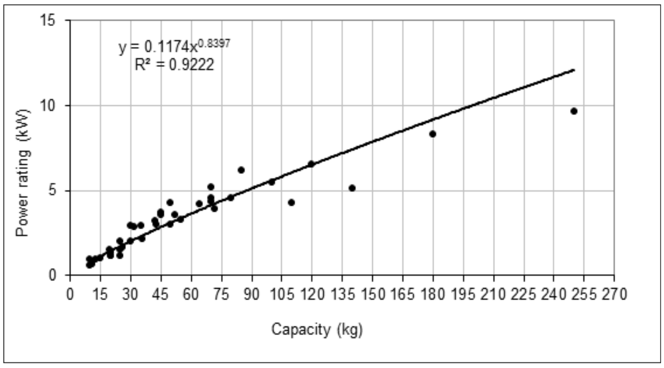


Figure 4 The relationship between the capacity and power rating of blast chillers estimated using data from manufacturers.

Table 7 Inventory data for the refrigerated storage (based on data from DEFRA (2008)).

	Number of days				
	1	2	3	4	5
Energy consumption ($kWh/kg_{cooked\ pasta}$)	0.006	0.012	0.019	0.025	0.031
Refrigerant load ($mg/kg_{cooked\ pasta}$)	5.48	10.96	16.44	21.92	27.40
Refrigerant leakage ($mg/kg_{cooked\ pasta}$)	0.82	1.64	2.47	3.29	4.11

Table 8 Inventory data for refrigerated transport.

	Fuel consumption (l/km)	Refrigerant charge (g/km)	Refrigerant leakage (g/km)
Truck 3.5-20 t	0.324	0.05	0.011
Truck 20-28 t	0.380	0.06	0.014
Truck> 28 t	0.417	0.07	0.016

Table 9 Inventory data for pasta regeneration (reheating) for different oven sizes.

	Small	Medium	Large
Combination oven – Gas			
Pre-heating energy ($kJ/kg_{cooked\ pasta}$)	10.96	7.35	6.02
Heating energy ($kJ/kg_{cooked\ pasta}$)	73.04	49.00	40.16
CO_2 ($g/kg_{cooked\ pasta}$)	5	3.16	2.59
CH_4 ($\mu g/kg_{cooked\ pasta}$)	84	56	46
N_2O ($\mu g/kg_{cooked\ pasta}$)	8	6	5
NOx ($mg/kg_{cooked\ pasta}$)	5	3	2
CO ($mg/kg_{cooked\ pasta}$)	2	2	1
Combination oven - Electric			
Pre-heating energy ($kJ/kg_{cooked\ pasta}$)	5.48	4.20	4.11
Heating energy ($kJ/kg_{cooked\ pasta}$)	36.52	28.00	27.39
Microwave oven			
Energy ($kJ/kg_{cooked\ pasta}$)	13.808		13.04

2.2.3 Cook-warm chain

In this chain, after the cooking stage, the food is transported to the point of use in insulated trucks (Figure 1). The Ecoinvent database has been used to estimates the impacts from the transport, making the same assumptions for the truck size and distances as for the refrigerated transport (see Section 2.2.2.2).

2.3 Sensitivity analysis

To test the robustness of the results and investigate the effect of key assumptions, the following parameters have been considered within the sensitivity analysis:

i) Pasta cooking

- the size of the pasta cookers and range tops: the capacity and power rating have been varied based on the respective relationships in Figure 3; note that the mean values are assumed in the base case; and
- emissions from fuel combustion: minimum and maximum emission factors for natural gas and LPG combustion defined by IPCC (2006) and EMEP/EEA (2013) have been considered, first by assuming all minimum and then all maximum values (see Table 1 and Table 2);

ii) Cook-chill and cook-warm chains

- the size of blast chillers (cook-chill): the capacity and power rating have been varied using the relationship in Figure 4; note that the mean values for power rating (6.5 kW) and capacity (120 kg) are assumed in the base case;
- refrigerant type for refrigerated storage (cook-chill): ammonia (R404A is assumed in the base case) and;
- refrigerant type for refrigerated transport (cook-chill): R134A and R410A (as above, R404A is assumed in the base case).
- the size of trucks: 3.5-20 t and >28 t, with 20-28 t assumed in the base case; and
- transport distance: 1 km and 100 km (50 km in the base case).

3 Results

The environmental impacts have been estimated using the midpoint ReCiPe method (Goedkoop et al., 2009). The following impact categories are considered: climate change (CC), ozone depletion (OD), human toxicity (HT), photochemical oxidants formation (POF), terrestrial acidification (TA), freshwater eutrophication (FE), terrestrial, freshwater and marine ecotoxicity (TE, FEc and ME, respectively), metal and fossil fuel depletion (MF and FD). Moreover, for the cooking operation only, the water footprint has also been estimated following the Pfister et al. methodology (2009).

SimaPro (V7.3.2) has been used for life cycle modelling and estimation of the impacts. The water footprint has been calculated using the CCaLC software tool (CCaLC, 2014).

The results are presented in the following sections, first for cooking in pasta cookers and the range tops, and then for the cook-chill and cook-warm chains.

3.1 Pasta cooking

As can be seen in
Figure 5, pasta cookers using natural gas are environmentally the best and electric cookers the worst option, with the difference between them ranging from 13% for fossil fuel depletion to 98% for freshwater eutrophication in favour of gas cookers. This is due to a relatively high contribution (21%) of coal and oil in the Italian electricity mix (based on 2011 data; Rapporto ISPRA, 2012; IEA, 2014). The exception is ozone depletion, for which the electric cookers are slightly better (by 2.5%) because of the emissions of halons used for fire retardants in gas pipelines. This impact is, on the other hand, highest for LPG cookers, being twice as high as for the electric appliances because of the production of offshore oil used in the life cycle of LPG. LPG cookers are also the worst option for freshwater ecotoxicity which is over 10 times higher than for the natural gas devices, owing to water discharge from the LPG production process.

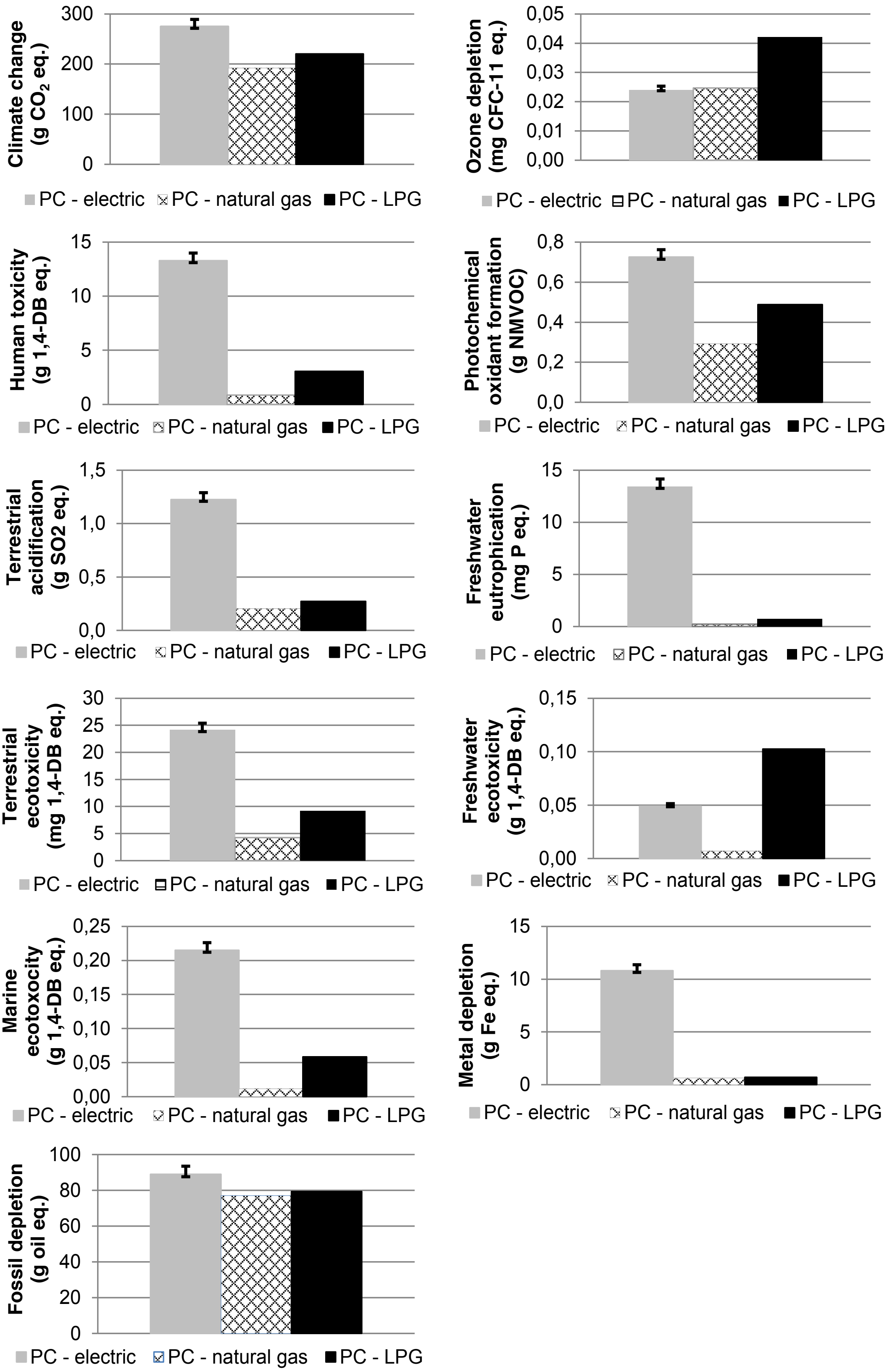


Figure 5 Environmental impacts of cooking in different types of pasta cooker (PC)

11

[All impacts expressed per 1 kg of cooked pasta, assuming the average size of pasta cookers estimated using the relationship in Figure 3. The height of the columns represent the average values. The error bars for electric pasta cookers represent the variation in impacts related to different size of the cookers. The error bars for the other two types of cooker not shown as the variation is <0.35%].

Figure 6 and Figure 7 compare pasta cookers and range tops using electricity and natural gas, respectively. As can be inferred from Figure 6, electric cookers are overall the best option compared to the electric range tops, with their impacts being on average 43% lower compared to the induction and 57% lower relative to the electric range tops. The latter appear to be environmentally least sustainable, while induction range tops represent the second best option after electric cookers, particularly when the lowest power rating is assumed.

Like the electric cookers, gas cookers also outperform gas range tops (Figure 7), with the savings in environmental impacts ranging from 34% for the climate change impact and ozone layer depletion to 66% for photochemical oxidants formation.

Varying the air emissions (see Table 1 and Table 2) from gas combustion for the gas-based equipment affects only three impact categories, as shown in Figure 8. While the overall effect on the climate change impact is small (~ 6%), terrestrial acidification and photochemical oxidant formation range widely (by ~130% and ~170%, respectively), with a much greater variation found for the gas than LPG devices. This is mostly due to NOx, which have a broader emissions range for natural gas than for LPG.

Therefore, based on the results of this study, it can be concluded that gas pasta cookers are the best option for most impacts, including the water footprint. The latter, given in Figure 9, is estimated at 0.75 l eq. per 1 kg of cooked pasta for pasta cookers, compared to 1.21 l eq. for the range tops.

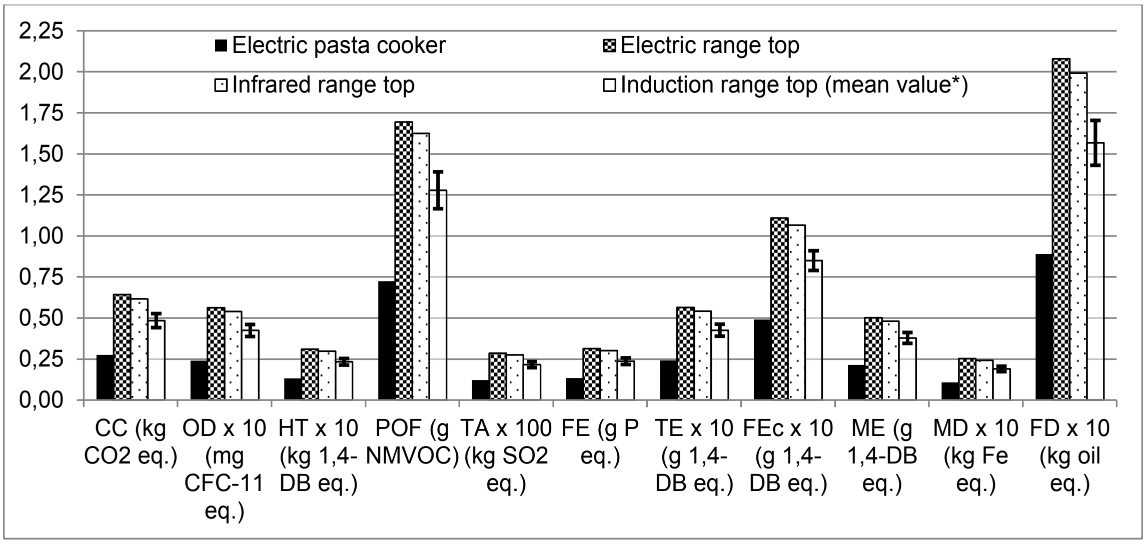


Figure 6 Comparison of environmental impacts of electric pasta cookers and range tops.
[All impacts expressed per 1 kg of cooked pasta. *Mean values represented by the height of the columns correspond to the average of the minimum and maximum power rating for induction range tops. The error bars show the impacts for the minimum and maximum power rating. The results for all other appliances correspond to the minimum power rating as explained in Section 2.2.1. Impacts nomenclature: CC: climate change; OD: ozone layer depletion; HT: human toxicity; POF: photochemical oxidant formation; TA: terrestrial acidification; FE: freshwater eutrophication; TE: terrestrial ecotoxicity; FEc: freshwater ecotoxicity; ME: marine ecotoxicity; MD: metal depletion; FD: fossil fuel depletion.]

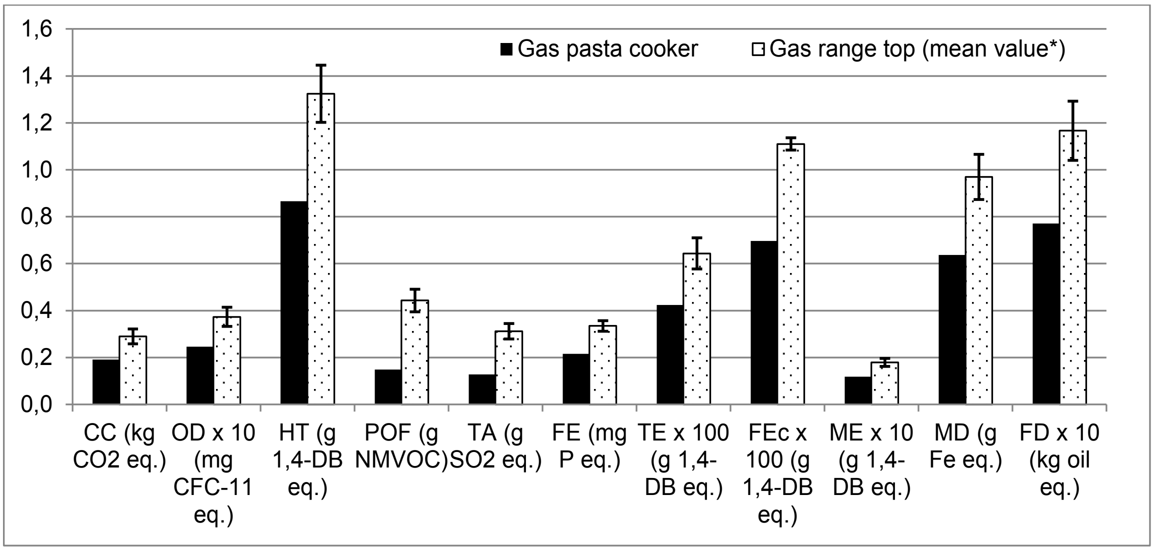


Figure 7 Comparison of environmental impacts of pasta cookers and range tops using natural gas.

[All impacts expressed per 1 kg of cooked pasta. For impacts nomenclature, see Figure 6. *Mean valued represented by the height of the columns correspond to the average of the minimum and maximum power rating for gas range tops. The error bars show the impacts for the minimum and maximum power rating. The results for gas pasta cookers correspond to the minimum power rating as explained in Section 2.2.1.]

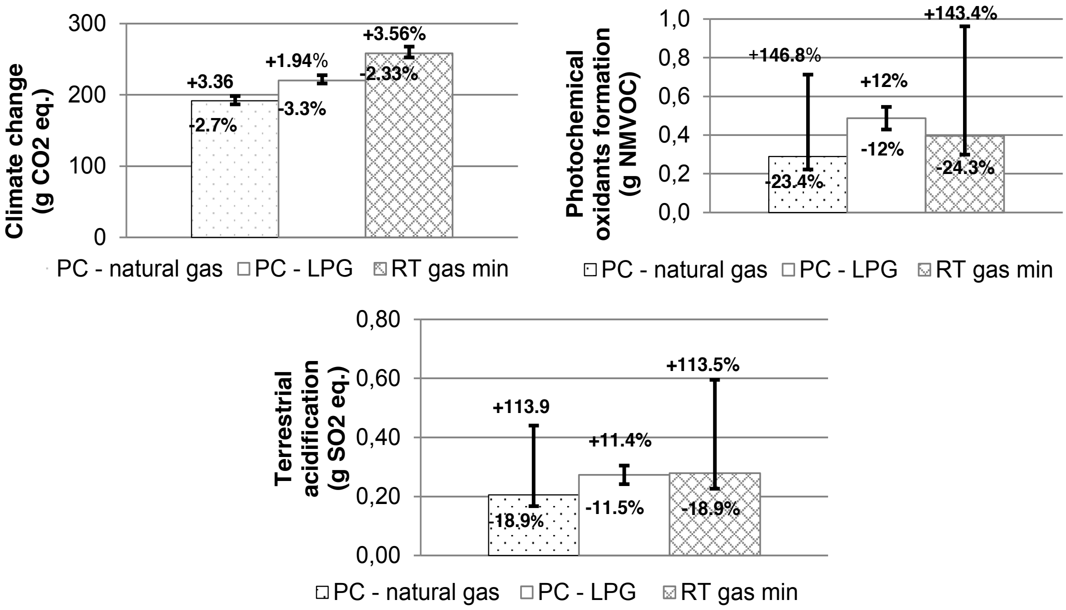


Figure 8 Environmental impacts of pasta cookers (PC) and range tops (RT) for the three categories affected by the changes in emissions from natural gas.

[All impacts expressed per 1 kg of cooked pasta. The height of the columns corresponds to the average air emissions from combustion of natural gas shown in Table 1 and Table 2. The error bars show the variation in the result assuming minimum and maximum values for the emissions. RT gas min: minimum power rating for gas range tops of 1.5 kW. NMVOC: non-methane volatile organic compounds.]

13

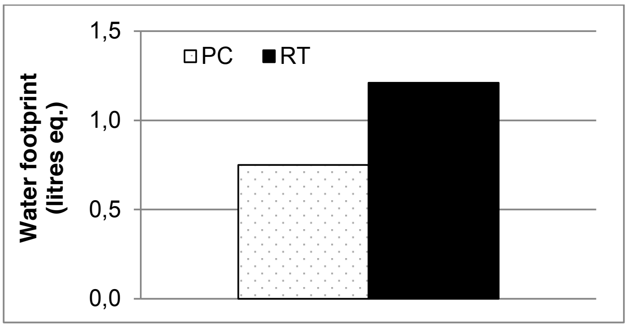


Figure 9 Water footprint of cooking for pasta cookers (PC) and range tops (RT).
[Expressed per 1 kg of cooked pasta. The water footprint considers only the amount of tap water required to cook pasta so that the impact is the same across the different types of pasta cookers and range tops, respectively.]

3.1.1 Comparison of results with literature

Only one study was found in the literature that considered the carbon footprint of pasta cooking in the catering sector (Barilla, 2013), estimating that 620 g CO_2 eq./kg of dry pasta is emitted when using gas appliances and 1300 g CO_2 eq. for electric devices (the exact type of appliances was not specified). This compares well with the GHG emissions of cooking estimated in the present study of 432 g CO_2 eq./kg of dry pasta for the gas cookers and the average value of 1307 g CO_2 eq./kg of dry pasta for the electric range tops. These values are equivalent to 192 g CO_2 eq. and 581 g CO_2 eq. per kg of cooked pasta, respectively, as presented in the previous section (see Figure 5 and Figure 6, respectively).

It is also interesting to put the results in perspective with respect to the contribution of pasta cooking to the impacts of the whole life cycle of pasta, when its production is also taken into account. There are several sources of data for the latter but they are mainly available for the carbon footprint and the values range widely, from 500-898 g CO_2 eq./kg of dry pasta (Federal Environment Agency, 2010; Röös et al., 2011; Barilla, 2013). Therefore, depending on the carbon footprint of pasta production considered, the contribution of cooking would range from 46% to 59%.

Only one study was found that considered impacts other than the carbon footprint (Barilla, 2013), estimating ozone depletion at 0.11 mg CFC11 eq./kg of dry pasta, acidification at 3.41 g SO_2 eq. and eutrophication at 4.82 g PO_4 eq. Based on these and the current study's results, the contribution of cooking to the life cycle of pasta (excluding the impacts from the cook-warm and cook-cold distribution) would be approximately 26% for ozone depletion and negligible for the other two impacts.

3.2 Cook-chill chain

The results for the cook-chill chain are presented in Figure 10, also showing the impacts of pasta cooking for context; as an example, the results are shown for pasta cookers. As can be observed from the figure, for most categories the contribution of cooking is much higher than of the other stages in the chain. This includes CC (67-77% of the total, depending on the pasta cooker used), TA (62-67%), FD (74-89%) and POF (64-72%). After cooking, blast chilling is the second highest contributor to the impacts, causing 18-19% of CC, 13-64% of FE, 12-47% of MD and 13-28% of TA, largely owing to the electricity used for chilling. The variation in the results is due to the different size of the chiller assumed (Figure 10), ranging from 0.81-12.11 kW as well as the different options in the cook-chill chain.

Unlike the other impacts, ozone depletion is largely due to blast chilling which contributes 73-87% to the total, with the rest being from cold storage of pasta. As this is due to the production of the refrigerant (R404A), a sensitivity analysis has been carried out to examine the effect on the results if ammonia is used instead for cold storage. The findings in Figure 11 suggest that the use of R404A leads to higher impacts for all the categories, except for TA which is lower for R404A by 7.7% because of the greater effect of ammonia leakage on this impact. The greatest variation is found for CC and OD which are 55% and more than 100 times higher, respectively, for R404A than ammonia. All other impact categories differ by less than 2%.

The contribution of refrigerated transport is small (0.02-7.5%) across the impact categories, except for POF to which is adds 14% to this impact for a distance of 50 km and 18% for 100 km. These findings are consistent with other food-related studies which also found that the contribution of refrigerated transport per functional unit is small (e.g. Eide, 2002; Fritsche and Eberle, 2009; Gunady et al., 2012). Nevertheless, to test the robustness of the results for transport, a sensitivity test has been performed assuming different sizes of trucks and the type of refrigerant used during transportation. The results in Figure 12 suggest that while the influence of the latter is negligible (<1%), the size of the truck affects the impacts of the transportation much more: they increase by 30-40% when a 3.5-20 t truck is used relative to the 20-28 t vehicle and decrease by up to 19% for a >28 t truck. The latter is due to bigger vehicles being more efficient, consuming less fuel per kilogram of product transported.

The effect of pasta regeneration (reheating) on the impacts is also small (0.06-4.5%). This appears to be in contrast with the findings by Schmidt Rivera et al. (2014) who identified reheating of a ready-made meal in an electrical oven as one of the hotspots in the life cycle. Moreover, in their analysis of the carbon footprint of bread, Espinoza-Orias et al. (2011) found toasting (effectively, reheating) to be one of the hotspots. These differences in the results could be explained by a much higher energy consumption for reheating assumed in these two studies because of the lower efficiency of domestic ovens and toasters compared to industrial ovens considered in the current work. Furthermore, unlike these studies, the current research assumes a full load of the ovens, thus further increasing the efficiency of energy consumption. Overall, the most environmentally efficient are gas ovens which are best for seven out of 11 impacts, followed by the microwave ovens with the lowest CC, OD, POF and FD (Figure 13). Electric ovens are the worst option across all the impact categories.

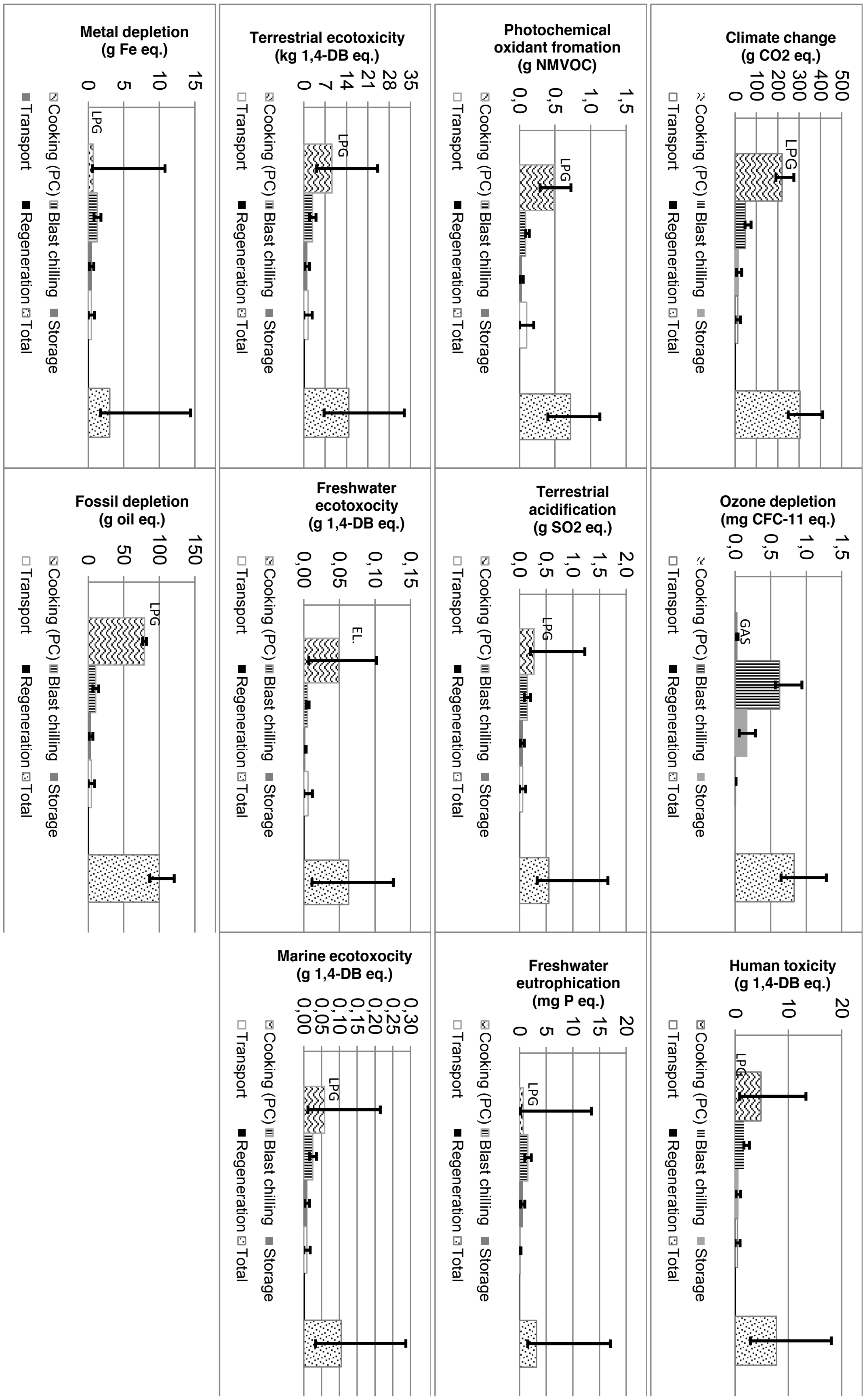
3.2.1 Comparison of results with literature

As there is a lack of studies related to the catering sector, it is not possible to compare the obtained results with literature. Nonetheless, some studies taking into account the cold chain for food products have been carried out. For example, Gunady et al. (2012) assessed the global warming potential (GWP) associated with the supply chain of three unprocessed foods which require refrigeration along their life cycle. They found that post-farm activities, which include packaging and refrigerated storage, accounted for 16-35% of the total GWP. Another study undertaken by Coley et al. (2009) indicated that the packing and refrigerated storage (and some administration activities) as responsible for approximately 24% of the total GHG emissions related to farm products. Even though the cited studies considered different kinds of product and life cycle stages (agriculture vs. processing) compared to the current study, there is a good agreement of the GWP results for the contribution of the 'cold stages' to the whole chain: in the present work, the blast chilling and cold storage are estimated to contribute on average 22% to the climate change impact (i.e. GWP).

3.3 Cook-warm chain

This chain, in addition to pasta cooking, comprises only one other stage – ambient transportation of pasta in insulated trucks; as the pasta is delivered warm to the consumption point, there is no need for reheating. The impacts are summarised in Figure 14, including cooking in pasta cookers, as for the cook-chill chain discussed in the previous section. Unsurprisingly, the majority of the impacts (79-100%) are from cooking with the contribution of transport being a little bit higher than in the cold chain, but still small: 0.09-9% across all the impact categories, except for POF to which is adds 16.7% for a distance of 50 km and 21.3% for 100 km. Many other studies of ambient transport of food have also found that this stage does not influence the impacts (e.g. Fusi et al., 2014; Espinoza-Orias et al., 2011).

The total impacts from the cook-warm and cook-chill chains are compared in the next section.


Climate change (g CO2 eq.)
Ozone depletion (mg CFC-11 eq.)
Human toxicity (g 1,4-DB eq.)
Photochemical oxidant fromation (g NMVOC)
Terrestrial acidification (g SO2 eq.)
Freshwater eutrophication (mg P eq.)
Terrestrial ecotoxicity (kg 1,4-DB eq.)
Freshwater ecotoxocity (g 1,4-DB eq.)
Marine ecotoxocity (g 1,4-DB eq.)
Metal depletion (g Fe eq.)
Fossil depletion (g oil eq.)
Cooking (PC)
Blast chilling
Storage
Transport
Regeneration
Total
LPG
GAS
EL.

16

Figure 10 Environmental impacts of the cook-chill chain.

[All impacts expressed per 1 kg of cooked pasta. The height of the columns and the error bars represent, respectively: for cooking, the type of pasta cooker as indicated in the figure with the impacts in between the best (minimum) and worst (maximum impact) cooker option assuming maximum power rating for all cookers; for blast chilling, the mean size (6.5 kW), minimum (0.81 kW) and maximum (12.11 kW); for refrigerated storage with R404A, the mean (3 days), minimum (1 day) and maximum (5 days) storage time; for transport by a 20-28 t truck with R404A refrigerant: the average (50 km), minimum (1 km) and maximum (100 km) distance considered; for regeneration: the average value for gas, electric combination and microwave ovens (no error bars).]

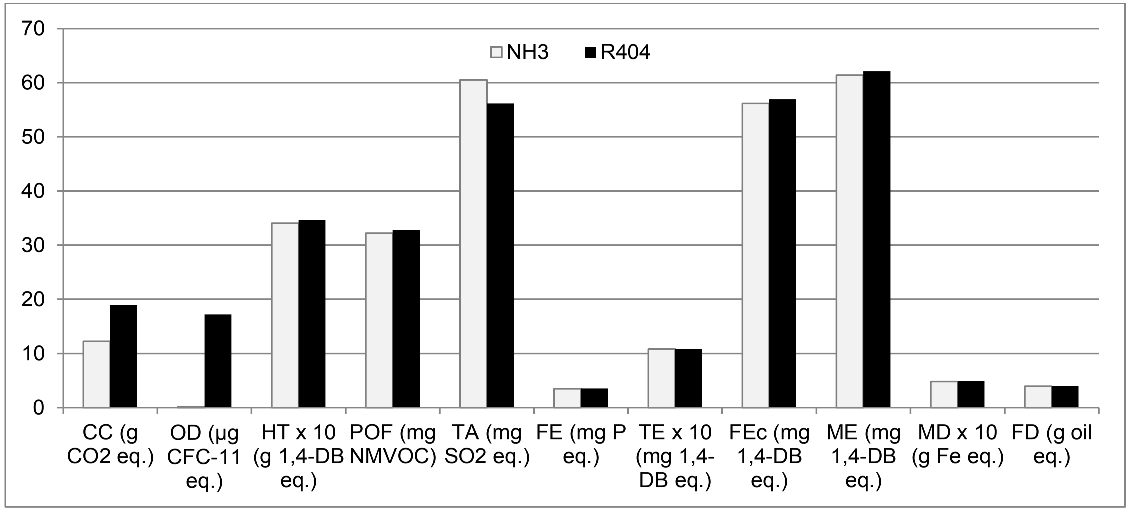
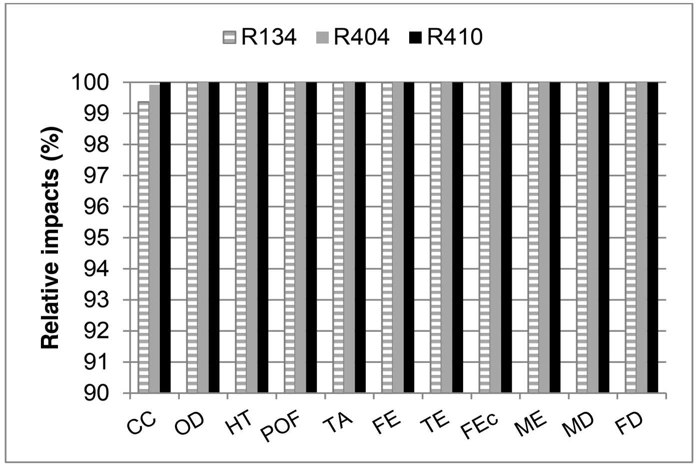
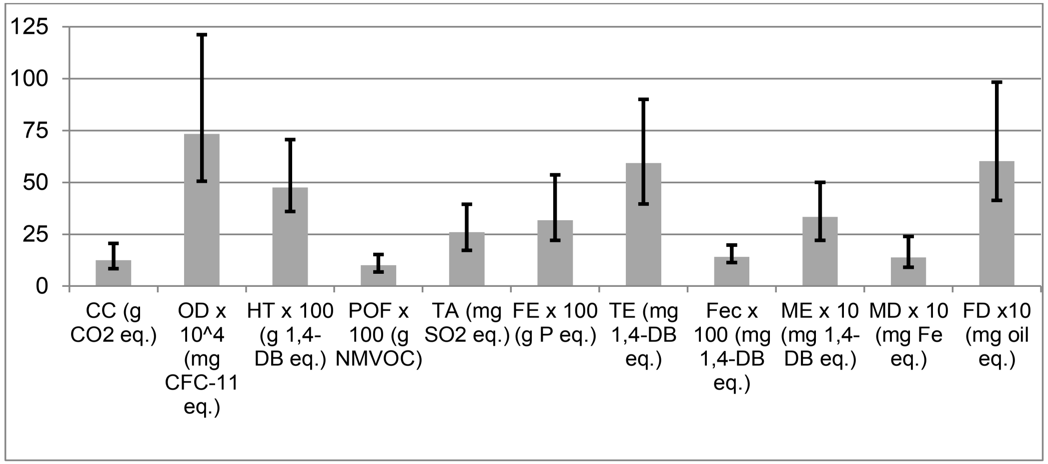


Figure 11 Comparison of the impacts for refrigerated storage using NH_3 and R404 as refrigerants.

[For impacts nomenclature, see Figure 6. The results refer to a three-day storage.]



a)



b)

Figure 12 The relative impacts of refrigerated transport using different refrigerants (a) and the effect on the impacts of the size of trucks (b).
[For impacts nomenclature, see Figure 6. The results refer to a distance of 50 km.]

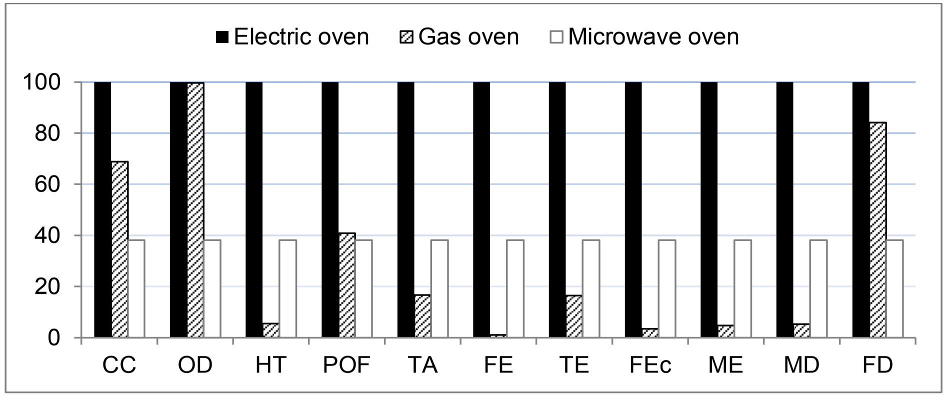


Figure 13 Relative impacts of different ovens for pasta regeneration (reheating).
[For impacts nomenclature, see Figure 6. The results refer to the mean oven size.]

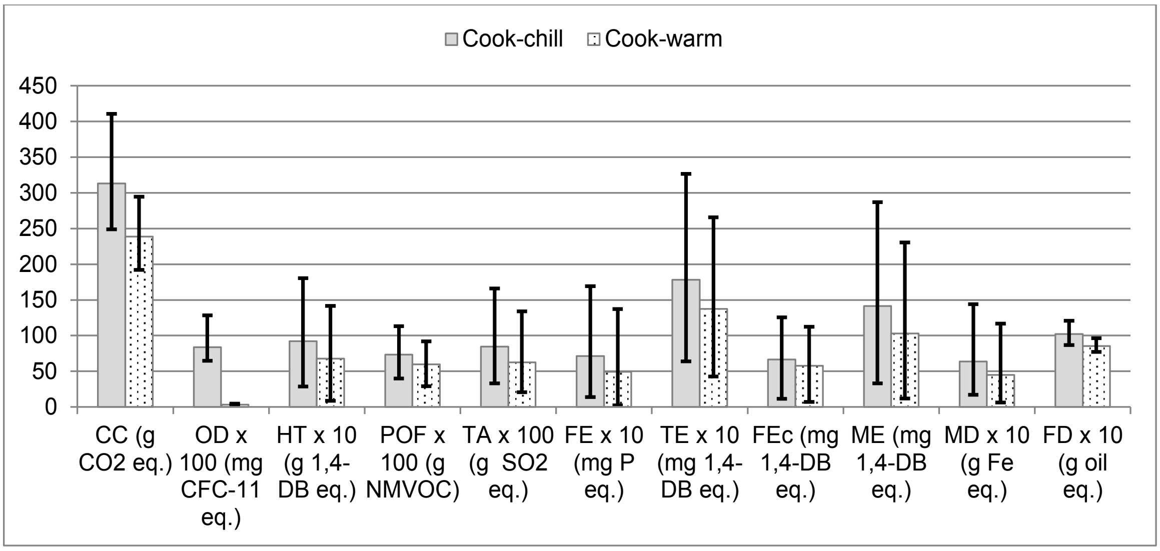


Figure 14 Comparison of the cook-chill and cook-warm chains
[For impacts nomenclature, see Figure 6. The height of the columns represents the mean values for all appliances and other parameters considered and the error bars show the minimum and maximum impacts based on the variations considered in the paper.]

3.4 Comparison of cook-chill and cook-warm chains

As indicated in Figure 14, all the impacts from the cook-warm chain are lower than from the cook-chill system, ranging from 17% and 30% lower FD and FE, respectively, to 96% lower OD.

Although neither chain is influenced by transportation, it is still interesting to compare the impacts from refrigerated and ambient transport used in the two respective chains. As expected, the environmental performance of the refrigerated transport is worse, particularly for CC owing to the increase in diesel fuel required for the refrigeration unit and the refrigerant leakage as well as OD because of the production of the refrigerant.

Therefore, the results of this study would suggest that the cook-warm chain is environmentally more sustainable than the cook-chill system. However, the latter tends to generate less food waste as only the amount of food which is actually required is reheated (Risteco, 2006a). According to a study carried out in some schools in Turin, Italy (Risteco, 2006b), the average percentage of first dishes (including pasta) not served, and therefore wasted, is 27.5%. Therefore, (possibly) avoiding waste through the adoption of the cook-chill chain, the impacts would be reduced because of the lower amount of pasta used and less waste that needs to be treated and disposed of. A similar conclusion was reached by Schmidt Rivera et al. (2014) in their study of ready-made meals, finding that the amount of waste is overall lower in the cold chain, leading to the lower overall impacts. Note that waste was not considered in this study as the impacts of pasta are not included in the system boundary, so that the inclusion of waste would not be congruent with the goal of the study.

Furthermore, the cook-chill chain provides more flexibility in terms of food preparation, allowing preparation of meals at any point in the day rather than just a few hours before the meal time, five days a week instead of seven (Risteco, 2006a). Moreover, the productivity tends to be higher in the cook-chill chain, with the number of meals prepared per day per chef being significantly greater (Clark, 1997). In addition, the cook-chill systems allow for wider menu choices with less skilled staff and reduced equipment needs (Smith and West, 2003). All these factors lead to increased efficiency and reduced costs, particularly labour (Clark, 1997; Risteco, 2006a; Marzano and Balzaretti, 2011).

Another important variable that should be taken into account when comparing different catering systems is the quality of meals delivered, both sensorial and nutritional. However, there are no conclusive findings on this with studies reporting conflicting results. For example, Light and Walker (1990) claim that the cook-hot-hold system results in damage to the quality of food, while Williams (1996) suggests that under normal operating conditions, with hot-holding limited to less than 90 minutes, vitamin retention is better than in a cook-chill chain. These aspects should therefore be investigated more fully in future research.

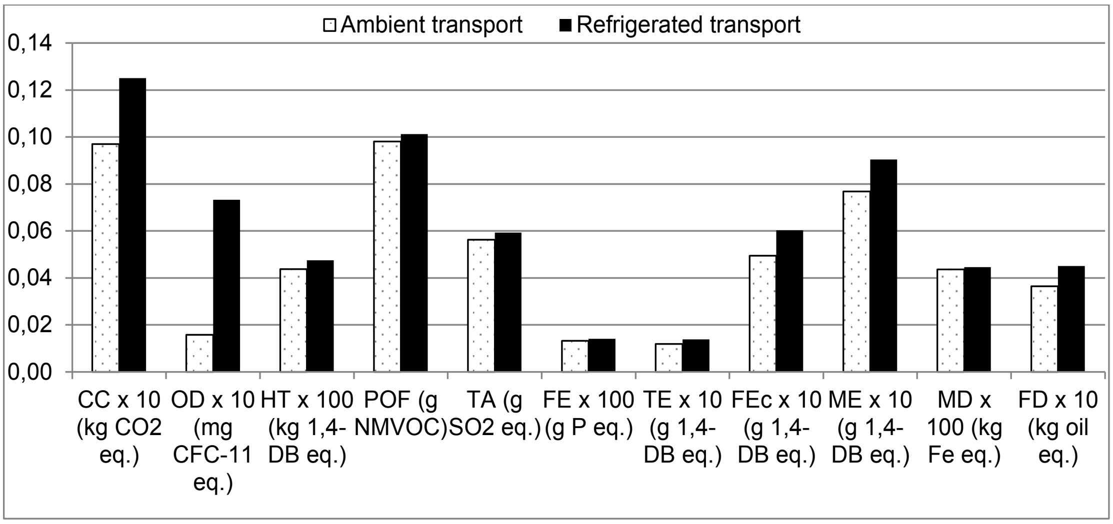


Figure 15 Comparison of ambient transport in the cook-warm and refrigerated transport in the cook-chill chains.
[All impacts expressed per 1 kg of cooked pasta. For impacts nomenclature, see Figure 6. The results refer to a distance of 50 km and a truck of 20-28 t.]

4 Conclusions

This work has studied different cooking technologies available in the food-service sector with the aim of identifying opportunities for improving its environmental performance. The focus of the study has been on pasta, one of the most popular foods worldwide. The following cooking methods have been considered: electric, gas and LPG pasta cookers and gas, electric, infrared and induction range tops. The second aim of the study has been the evaluation of the environmental impacts of different deferred systems available in the food-service sector, namely the cook-chill and cook-warm chains.

The results suggest that cooking in pasta cookers saves up to 60% of energy and 38% of water compared to range tops and therefore reduces by 34-66% the impacts associated with pasta preparation. Nevertheless, it should be noted that pasta cookers and range tops serve different purposes: the latter are manly used for preparation of smaller meal quantities, making them suitable for à-la-carte business in restaurants or in hospital kitchen; pasta cookers, on the other hand, are used when much larger amounts of food need to be cooked.

The environmental impacts of pasta cooking could also be reduced by using gas rather than electric appliances as the impacts of the latter are higher by 13-98%. A further improvement would be achieved by using a lid on the cooking appliances. However, their use in professional kitchens is not a regular practice, for both cost reasons (lids are sold as an optional equipment for pasta cookers) and for the convenience of cooking staff.

Pasta cooking is the major contributor to the environmental impacts in both the cook-chill and cook-warm chains. In the former, blast chilling is the main cause of ozone depletion and the second highest contributor to all other impacts. The contribution of refrigerated transport and storage is small, except for photochemical oxidant formation, for which the former contributes 14-18%, depending on the distance considered. The ambient transport used in the cook-warm chain influences photochemical oxidant formation, contributing 17-21% to the total.

Overall, the results of this work indicate that the cook-chill chain has 17-96% higher environmental impacts than the cook-warm system. This is mainly due to the use of refrigerants and higher consumption of energy. Therefore, the cook-warm approach appears to be environmentally a more sustainable option under the conditions considered in this study.

However, the choice of the 'best' chain would depend on many other factors, including flexibility, efficiency, costs, convenience and food quality, the consideration of which was beyond the scope of this paper. It is therefore recommended that these parameters be considered in future studies.

Acknowledgements
This work has been funded by the European Commission under the Socrates Exchange Programme and the UK Engineering and Physical Sciences Research Council (EPSRC), Grant no. EP/K011820. This funding is gratefully acknowledged.

Appendix

Table A.1 Water loss for different pasta cookers

	Water losses (kg/kg cooked pasta)	
	Electric cooker	**Gas/LPG cooker**
Water absorbed by pasta	0.550	0.550
Water associated with foam (generated during cooking)	0.097	0.097
Water vapour	0.220	0.180
Water loss while draining pasta (5%)	0.220	0.220
Total loss (refill for the 2nd cycle)	**1.087**	**1.047**

Table A.2 Data used for calculating energy requirements for water heating for different pasta cookers

	Electric cooker		**Gas/LPG cookers**	
	Temperature (°C)	**Mass (kg/kg cooked pasta)**	**Temperature (°C)**	**Mass (kg/kg cooked pasta)**
Water reused from the 1st cycle	100	3.353	100	3.393
Water lost in the 1st cycle and toped up in the 2nd cycle (see Table A.1)	14.5	1.087	14.5	1.047
Total water (reused and refilled water)	78.9	4.440	79.7	4.440

Table A.3 Distribution of water, heating energy and wastewater between the 1st and 2nd cycle for different pasta cookers

	Electric pasta cookers			**Gas/LPG**		
	1st cycle	**2nd cycle**	**Average**	**1st cycle**	**2nd cycle**	**Average**
Water use (kg/kg cooked pasta)	4.440	1.087	2.764	4.440	1.047	2.744
Heating energy (MJ/kg cooked pasta)	1.633	0.403	1.018	3.182	0.756	1.969
Wastewater (kg/kg cooked pasta)	0.317[a]	3.670[b]	1.990	0.317[a]	3.710[b]	2.014

[a]The sum of water associated with the foam (discharged to the drain) and water loss while draining pasta (see Table A.1)

[b]Total amount of water for pasta cooking (4.44 kg) minus the amount absorbed by pasta (0.55 kg) and lost through evaporation (0.22); see Table A.1 for the latter two values.

The initial temperature of water in the 2nd cycle is calculated as follows:

$$T_t = \frac{W_1\, x\, T_1 + W_r\, x\, T_r}{W_t} \quad (°C) \tag{A.1}$$

where:

T_t temperature of the water in the 2nd cycle (°C)
T_1 temperature of W_1 (°C)
W_1 mass of water reused from the 1st cycle (kg)
W_r mass of water refilled for cooking pasta in the 2nd cycle (kg)
T_r temperature of W_r (°C)
W_t sum of reused and refilled water (kg)

5 References

ANGEM, 2014. Ristorazione collettiva in sintesi Associazione Nazionale delle Aziende di Ristorazione Collettiva e Servizi Vari. www.angem.it/it.

Baldwin C., Wilberforce N., Kapur A., 2011. Restaurant and food service life cycle assessment and development of a sustainability standard. Int J LCA, 16: 40-49.

Barilla, 2013. Environmental product declaration of durum wheat semolina dried pasta for 5 kg food-service in catering packaging. www.environdec.com.

Bevilacqua M., Braglia M., Carmignani G., Zammori F.A., 2007. Life cycle assessment of pasta production in Italy. J Food Quality, 30(6):932 – 952.

Bourlakis M. A., Weightman P.W.H., 2004. Food supply chain management, Chapter 1. Blackwell Publishing.

Bovea M. D., Cabello R., Querol D., 2007. Comparative life cycle assessment of commonly used refrigerants in commercial refrigeration systems. Int J LCA 12 (5): 299 – 307.

Buonanno G., Stabile L., Morawska L., 2009. Particle emission factors during cooking activities. Atmospheric Environment, 43(20): 3235-3242.

CCaLC (2014). CCaLC software and database V3.2. www.ccalc.org.uk.

Ciappellano S., 2009. Manuale della ristorazione. Casa Editrice Ambrosiana, C.E.A., Milano.

Clark J.R., 1997. Improving catering productivity by using cook-chill technology. The Cornell Hotel and Restaurant Administration Quarterly, 38(6): 60-67.

Coley D., Howard M., Winter M., 2009. Local food, food miles and carbon emissions: A comparison of farm shop and mass distribution approaches. Food Policy, 34: 150–155.

Davis B, Lockwood A, Stone S (1998). Food and beverage management, 3rd ed. Butterworth-Heinemann: Oxford.

Decreto del Presidente della Repubblica (D.P.R.) 19 ottobre 2011, n. 227. Regolamento per la semplificazione di adempimenti amministrativi in materia ambientale gravanti sulle imprese, a norma dell'articolo 49, comma 4-quater, del decreto-legge 31 maggio 2010, n. 78, convertito, con modificazioni, dalla legge 30 luglio 2010, n. 122. (12G0013). Gazz. Uff. 3 febbraio 2012, n. 28.

DEFRA, 2008. Greenhouse Gas Impacts of Food Retailing. Defra Research Project FO 0405. Department for the Environment Food and Rural Affairs, London. www.defra.gov.uk.

Duiven, J. E., Binard, P., 2002. Refrigerated storage: new developments. Bulletin of the IIR [2002-2]. www.iifiir.org.

Eide M.H., 2002. Life cycle assessment (LCA) of industrial milk production. Int J LCA, 7(2): 115-126.

EMEP/EEA, 2013. Emission inventory guidebook 2013, Part B, 1.A.4 (Small combustion). EEA (European Environment Agency), 29/08/2013. www.eea.europa.eu/publications/emep-eea-guidebook-2013.

EPA, 2013. United States Environmental Protection Agency. www.epa.gov.

EpiCentro, 2012. Guida per il miglioramento della qualità nutrizionale nella ristorazione scolastica. Centro Nazionale di Epidemiologia, Sorveglianza e Promozione della Salute. www.epicentro.iss.it/default.asp.
Espinoza-Orias N., Stichnothe H., Azapagic A., 2011. The carbon footprint of bread. Int J LCA, 16:351–365.
European Committee For Standardization (CEN), 2008. UNI EN 30-1-3:2008. Domestic cooking appliances burning gas - Part 1-3: Safety - Appliances having a glass ceramic hotplate. CEN: Brussels.
European Committee For Standardization (CEN), 2005. EN 203-2-1:2005. Gas heated catering equipment - Part 2-1: Specific requirements - Open burners and wok burners. CEN: Brussels.
Evans J., Russell S., James S., 1996. Chilling of recipe dish meals to meet cook-chill guidelines. Int J of Refrig, 19(2): 79-86.
Food Service Europe (FERCO), 2014. European Industry overview. www.foodserviceeurope.org/.
Food Service Technology Center, 2002. FSTC Report 5011.02.26, Commercial cooking appliance technology assessment. Fisher-Nickel Inc., 12949 Alcosta Boulevard, Suite 101 San Ramon, CA 94583.
Food Service Technology Center, 1999. FSTC Report 5011.99.53, Development and validation of a standard test method for pasta cookers. Pacific Gas & Electric Company, San Francisco California.
Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Heck, T., Hellweg, S.,Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G., 2007. Overview and methodology. Ecoinvent report No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf.
Fritsche U.R., Eberle U., 2009. Greenhouse-gas emissions from the production and processing of food - Working paper. Öko-Institut e.V, Darmstadt, July 2009.
Fusi, A., Guidetti, R., Benedetto, G., 2014. Delving into the environmental aspect of a Sardinian white wine: from partial to total life cycle assessment. Science of the Total Environment, 472:989-1000.
Goedkoop, M.J., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., Van Zelm, R., 2009. ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level, First edition, Report I: Characterisation; 6 January 2009. www.lcia-recipe.net.
Gunady M.G.A., Biswas W., Solah V.A., James A.P., 2012. Evaluating the global warming potential of the fresh produce supply chain for strawberries, romaine/cos lettuces (Lactuca sativa), and button mushrooms (Agaricus bisporus) in Western Australia using life cycle assessment (LCA). J Cleaner Prod. 28: 81-87.
IEA, 2014. IEA Statistics. International Energy Agency. www.iea.org/statistics/statisticssearch/report/?country=ITALY=&product=electricityandheat&year=Select.
International Organization for Standardization (ISO), 2006a. ISO 14040. Environmental management - life cycle assessment - principles and framework. ISO, Geneva.
International Organization for Standardization (ISO), 2006b. ISO 14044. Environmental management - life cycle assessment - requirements and guidelines. ISO, Geneva.
Intergovernmental Panel on Climate Change (IPCC), 2006. IPCC guidelines for national greenhouse inventories for energy. Chapter 2: Stationary combustion, Vol. 2, IPCC.
Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA), 2012. Fattori di emissione di CO_2 nel settore elettrico e analisi delle decomposizione delle emissioni. Rapporto 172/2012.
Light N., Walker A., 1990. Cook-chill catering – technology and management, Ch. 1. Elsevier Science Publishers: New York.
Marti A., Seetharaman K., Pagani M.A., 2013. Rheological approaches suitable for investigating starch and protein properties related to cooking quality of durum wheat pasta. J Food Quality, 36: 133–138.
Marzano M.A., Balzaretti C.M., 2011. Cook-serve method in mass catering establishments: Is it still appropriate to ensure a high level of microbiological quality and safety? Food Control, 22: 1844-1850.
Metropolitana Milanese SPA, 2012. Servizio idrico: acquedotto. www.metropolitanamilanese.it/pub/page/it/MM/acquedotto_distribuzione.

Museo energia, 2013. Cottura dei cibi e ricadute energetiche e ambientali. www.museoenergia.it/museo.php.
Federal Environment Agency, 2010. ProBas Database, Federal Environment Agency, Germany. www.probas.umweltbundesamt.de/php/index.php.
Pfister S., Koehler A., Hellweg S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environmental Science & Technology 43 (11): 4098-4104.
Risteco, 2006a. Eco-efficienza nella ristorazione moderna. Quaderno 3: La logistica. www.ecomeal.info/?g=innover.
Risteco, 2006b. Eco-efficienza nella ristorazione moderna. Quaderno 4: La gestione della mensa. www.ecomeal.info/?g=innover.
Rohatsch M., Lemme F., Neumann P., Wagner F., 2007. Professional Kitchens, Planning Design Equipment. FCSI Europe - Africa - Middle East, HUSS-MEDIEN.
Röös E., Sundberg C., Hansson P., 2011. Uncertainties in the carbon footprint of refined wheat products: a case study on Swedish pasta. Int J LCA, 16: 338–350.
Ruini L., Ferrari, E., Meriggi P., Marino M., Sessa F., 2013a. Increasing the sustainability of pasta production through a life cycle assessment approach. Report, 4th International Workshop Advances in Cleaner production. São Paolo, Brazil, 22nd – 24th May 2013.
Ruini L., Marino M., Pignatelli S., Laio F., Ridolfi L., 2013b. Water footprint of a large-sized food company: The case of Barilla pasta production. Water Resources and Industry, 1–2: 7–24.
Schmidt Rivera X., Orias N.E., Azapagic A. Life cycle environmental impacts of convenience food: comparison of ready and home-made meals. J Cleaner Prod., 73: 294–309.
Smith A. G., West A., 2003. Catering systems. Catering Elsevier Science: Oxford.
UNEP, 2003: 2002 Report on the refrigeration, air conditioning and heat pumps technical options committee – 2002 assessment. UNEP Ozone Secretariat, Nairobi, Kenya.
Wallén A., Brandt N., Wennersten R., 2004. Does the Swedish consumer's choice of food influence greenhouse gas emissions? Environmental Science and Policy 7: 525–535.
Williams P. G., 1996. Vitamin retention in cook/chill and cook/hot-hold hospital foodservices, J American Dietetic Association, 96(5): 490-498.
Wolf M.A., Pant R., Chomkhamsri K., Sala S., Pennington D., 2012. The international reference life cycle data system (ILCD) handbook - towards more sustainable production and consumption for a resource-efficient Europe. JRC Reference Reports. http://publications.jrc.ec.europa.eu/repository/handle/111111111/25589.
Zhang Q., Gangupomu R.H., Ramirez D., Zhu Y., 2010. Measurement of ultrafine particles and other air pollutants emitted by cooking activities. Int J Environ. Res. Public Health, 7:1744-1759.

7. Conclusions

The case studies presented in this work provide a clear demonstration of how LCA can be used on agro-food products to identify environmental hotspots and therefore formulate possible ameliorative solutions, which can be in turn evaluated though the LCA in order to quantify the benefits arising from their application.

The impact of the various stages of food's life cycle (agriculture, processing, packaging, distribution, use and disposal) is not constant and strongly depend on the product taken under consideration. In the case of wine for example, the most impactful phase is the production of the packaging material. Given this result, it is possible to formulate some ameliorative options, such as using a lighter glass bottle or substituting the glass bottle with an aseptic carton. In the case of fresh-cut salad instead, the impact of packaging material is almost negligible; the main hotspot is represented by the processing phase due to the high consumption of energy and the use of water and related production of wastewater, which need to be treated. As the water consumed during the processing phase is responsible for a large environmental impact, a possible ameliorative solution would be the installation of a filtration plant for the recovery of part of the washing solution. This option was evaluated in order to quantify the environmental benefits arising from its application; based on the results, it could be concluded that the reuse of processing water represents an effective solution for improving the environmental performance of ready-to-eat vegetables.

When a single phase of a product's life cycle was evaluated, different technical solutions were proposed and assessed in order to identify the most environmentally sustainable one. In the case of cereals, different cropping systems as well as seedbed preparation techniques have been assessed: results show that the single crop is more environmentally efficient with respect to double crop system, while different seedbed techniques do not produce significant changes in the environmental impact of cereals cultivation. The case study undertaken on pasta in the catering sector has identified instead the most sustainable options for pasta cooking and quantified the difference in terms of environmental impacts between two deferred catering systems.

In the light of increasing the sustainability of food production, it is extremely important, on the one hand, to measure the impacts produced by food products and identify the most critical phases on which intervene to obtain relevant improvements and, on the other hand, to quantify such improvements, assessing different possible alternative solutions.

8. References

Aranda A., Scarpellini S., Zabalza I., 2005. Economic and environmental analysis of the wine bottle production in Spain by means of life cycle assessment. International Journal of Agricultural Resource Govern Ecol, 4: 178-191.

Ardente F., Beccali G., Cellura M., Marvuglia A., 2006. POEMS: a case Study of an italian wine-producing firm. Environmental Management, 3 : 350–364.

Benedetto G., 2013. The environmental impact of a Sardinian wine by partial Life Cycle Assessment. Wine Economics and Policy, 2: 33-41.

Berlin J., 2002. Environmental life cycle assessment (LCA) of Swedish semi-hard cheese. International Dairy Journal, 12: 939–953.

Bevilacqua M., Braglia M., Carmignani G., Zammori F.A., 2007. Life Cycle Assessment Of Pasta Production In Italy. Journal of Food Quality, 30: 932–952.

Blengini G.A., Busto M., 2009. The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy), Journal of Environmental Management, 90: 1512–1522.

Braschkat J., Patyk A., Quirin M., Reinhardt G.A., 2003. Life cycle assessment of bread production - a comparison of eight different scenarios. (Paper presented at the 4th Conference on LCA in the Agri-Food Sector, Bygholm, Denmark).

Brentrup F., Küsters J., Lammela J., Barraclough P., Kuhlmann H., 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. European Journal of Agronomy, 20: 265–279.

Brundtland Commission, 1987. Report of the World Commission on Environment and Development. United Nations.

Casey J.W., Holden N.M., 2003. A systematic description and analysis of GHG emissions resulting from Ireland's milk production using LCA methodology. In: Proceedings of the Fourth International Conference on Life Cycle Assessment in the Agri-Food Sector, Bygholm, Denmark.

Charles R., Jolliet O., Gaillard, G., Pellet D., 2006. Environmental analysis of intensity level inwheat crop production using life cycle assessment. Agriculture, Ecosystems and Environment, 113: 216–225.

Cederberg C., Mattsson B., 2000. Life cycle assessment of milk production – a comparison of conventional and organic farming. Journal of Cleaner Production, 8 (1): 49–60.

Cordella M., Tugnoli A., Spadoni G., Santarelli F., Zangrando T., 2008. LCA of an Italian Lager Beer. The International Journal of Life Cycle Assessment, 13: 133-139.

Cowell SJ., 1998. Environmental Life Cycle Assessment of Agricultural Systems: Integration IntDecision-Making. PhD thesis. University of Surrey, Guildford, UK.

Denman, K. et al. 2007. Couplings Between Changes in the Climate System and Biogeochemistry - IPCC, Climate Change 2007: The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. Averyt, M. Tignor and H. Miller (eds.)], Cambridge University Press, Cambridge, United Kingdom and United States.

Dreyer L. C., Niemann A. L., Hauschild M. Z., 2003. Comparison of three different LCIA methods: EDIP97, CML2001 and Eco-indicator 99. Does it matter which one you choose? The International Journal of Life Cycle Assessment, 8(4): 191-200.

Eide, M.H., 2002. Life cycle assessment of industrial milk production. http://www2.lib.chalmers.se/cth/diss/doc/0102/HogaasEideMerete.html.

Espinoza-Orias N., Stichnothe H., Azapagic A., 2011. The carbon footprint of bread. International Journal of Life Cycle Assessment, 16: 351–365.

FAO, 2013. Food Wastage Footprint: Impacts on Natural Resources. Summary Report. Food and Agriculture Organisation, Rome. www.fao.org/docrep/018/i3347e/i3347e.pdf

Gazulla C., Raugei M., Fullana-i-Palmer P., 2010. Taking a life cycle look at crianza wine production in Spain: where are the bottlenecks? Int J Life Cycle Assess, 15: 330–337.

Gonzalez A., Klimchuk A., Martin M., 2006. Life Cycle Assessment of Wine production Process. Finding Relevant Process Efficiecy and Comparison to Eco-Wine Production, 1N1800 Life Cycle Assessment, Group 4.

González-García S., Castanheira E. G., Dias A. C., Arroja L., 2013a. Environmental life cycle assessment of a dairy product: the yoghurt. Int J Life Cycle Assess, 8: 796–811.

González-García S., Castanheira E. G., Dias A. C., Arroja L., 2013b. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. Science of The Total Environment, 442: 225–234.

Guinée J.B., Gorrée M., Heijungs R., Huppes G., Kleijn R., Koning A. de, Oers L. van, Wegener Sleeswijk A., Suh S., Udo de Haes H.A., Bruijn H. de, Duin R. van, Huijbregts M.A.J., 2002. Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Kluwer Academic Publishers, ISBN 1-4020-0228-9, Dordrecht, 2002, 692 pp.

Harada H., Kobayashi H., Shindo H., 2007. Reduction in greenhouse gas emissions by no-tilling rice cultivation in Hachirogata polder, northern Japan: Life-cycle inventory analysis, Soil Science and Plant Nutrition, 53: 668–677.

Hospido A., Moreira M.T., Feijoo G., 2003. Simplified life cycle assessment of Galician milk production. International Diary Journal, 13 (10): 783–796.

Hospido, A., Moreira, M.T., Feijoo, G., 2005. Environmental analysis of beer production. International Journal of Agricultural Resources, Governance and Ecology, 4 (2): 152–162.

International Organization for Standardization (ISO), 2006. ISO 14025. Environmental labels and declarations -- Type III environmental declarations -- Principles and procedures.

International Organization for Standardization (ISO), 2006. ISO 14040. Environmental Management - Life Cycle Assessment - Principles and Framework.

International Organization for Standardization (ISO), 2006. ISO 14041. Environmental Management - Life Cycle Assessment - Goal and scope definition and inventory analysis.

International Organization for Standardization (ISO), 2006. ISO 14042. Environmental Management - Life Cycle Assessment – Life Cycle Impact Assessment.

International Organization for Standardization (ISO), 2006. ISO 14043. Environmental Management - Life Cycle Assessment – Life Cycle Interpretation.

International Organization for Standardization (ISO), 2006. ISO 14044. Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

International Organization for Standardization (ISO), 2014. ISO 14046. Environmental Management -Water footprint - Principles, requirements and guidelines.

International Organization for Standardization (ISO), 2013. ISO 14067. Greenhouse gases - Carbon footprint of products - Requirements and guidelines for quantification and communication.

Joint Research Centre, European Commission, 2010. ILCD Handbook: General Guide for Life Cycle Assessment – Detailed Guidance. Italy, 2010.

Joint Research Centre, European Commission, 2011. ILCD Handbook: Recommendations for Life Cycle Impact Assessment in the European context. Italy, 2011.

Koroneos C., Roumbas G., Gabari Z., Papagiannidou E., Moussiopoulos N., 2005. Life cycle assessment of beer production in Greece. Journal of Cleaner Production, 13 (4): 433–439.

Meadows D.H., Meadow D.L., Randers J., Beherens III W., 1972. The limits to growth - A report for

the club of rome's project on the predicament of mankind. Universe Book, New York.

Meisterling K., Samaras C., Schweizer V., 2009. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. Journal of Cleaner Production, 17: 222–230.

Neto B., Dias A.C., Machado M., 2013. Life cycle assessment of the supply chain of a Portuguese wine: from viticulture to distribution. Int J Life Cycle Assess, 18: 590–602.

Notarnicola B., Nicoletti G.M., 2001. Life cycle assessment of pasta and couscous. Tecnica Molitoria, 52(1): 19-27, 33.

Notarnicola B., Mongelli I., Tassielli G., Nicoletti, G.M., 2004. Environmental Input-Output Analysis and Hybrid Approaches to improve the set up of pasta life Cycle Inventory. Journal of Commodity Science, Technology and Quality 2004, 43(III): 59-86.

Remmen A., Jensen A., Frydendal J., 2007. Life cycle management: a business guide tosustainability. Nairobi, Kenya: UNEP/SETAC.

Roy P., Ijiri T., Nei D., Orikasa T., Okadome H., Nakamura N., Shiina T., 2009. Life cycle inventory (LCI) of different forms of rice consumed in households in Japan. Journal of Food Engineering, 91: 49–55.

Roy P., Shimizu, N., Kimura T., 2005. Life cycle inventory analysis of rice produced by localprocesses. Journal of the Japanese Society of Agricultural Machinery 67, (1): 61–67.

Roy P., Shimizu N., Okadome H., Shiina T., Kimura T., 2007. Life cycle of rice: Challenges and choices for Bangladesh. Journal of Food Engineering, 79: 1250–1255.

Rugani B., Niccolucci, V., Pulselli R.M., Tiezzi E., 2009. A cradle-to-gate Life Cycle Assessment integrated with Emergy evaluation: sustainability analysis of an organic wine production. In: Proceedings of the SETAC Europe 19th Annual Meeting on Protecting Ecosystem Health: Facingthe Challenge of a Globally Changing Environment, May 31st–June 4th, 2009, Göteborg, Sweden, p. 274.

Ruini L., Ferrari E., Meriggi P., Marino M., Sessa F., 2013. Increasing the Sustainability of Pasta Production Through a Life Cycle Assessment Approach. 4th int workshop in advanvces in cleaner prod 2013.

Salomone R., Ioppolo G., 2012. Environmental impacts of olive oil production: a Life Cycle Assessment case study in the province of Messina (Sicily). Journal of Cleaner Production, 28: 88-100.

Schmidt Rivera X., Orias N.E., Azapagic A. Life cycle environmental impacts of convenience food: comparison of ready and home-made meals. J Cleaner Prod, 73: 294–309.

Sonnesson U., Belin J., Ziegler F., 2010. Environmental Assessment and Management in the Food Industry – Life Cycle Assessment and Related Approaches. Elsevier Book, 2010.

Soubbotina T., 2004. Beyond Economic Growth. WBI Learning Resources Series, The World Bank, Washington D.C.

Steinfeld H., Gerber P., Wassenaar T., Castel V., Rosales M., de Haan C., 2006. Livestock's long shadow,. FAO, Rome 2006.

Takamoto Y., Mitani Y., Takashio M., Itoi K., Muroyama K., 2004. Life Cycle Inventory Analysis of a Beer Production Process. MBAA TQ,, 41(4): 363–365.

Tukker A., Huppes G., Guinée J., Heijungs R., de Koning A., van Oers L., Suh S., Geerken T., VanHolderbeke M., Jansen B., Nielsen P., 2006. Analysis of the LifeCycle Environmental Impacts Relatedto the Final Consumption of the EU-25. Main report. IPTS/ESTO project. http://ec.europa.eu/environment/ipp/pdf/eipro_report.pdf.

United Nations, 2000. Millennium Declaration. http://www.un.org/millennium/declaration/ares552e.htm.

United Nations Development Programme (UNDP), 1996. Human Development Report. Oxford University Press, 1996.

Vázquez-Rowe I., Villanueva-Rey P., Moreira M.T., Feijoo G., 2012. Environmental analysis of Ribeiro wine from a timeline perspective: Harvest year matters when reporting environmental impacts. J. Environ. Manage, 98: 73-83.

Williams A.G., Audsley E., Sandars D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. In: Main Report, Defra Research Project IS0205, Cranfield University and Defra. http://www.defra.go.uk.

World Commission on Environment and Development, 1987. Report: Our Common Future. http://www.un-documents.net/our-common-future.pdf.

9. Acknowledgments

Grazie a tutti i Professori coinvolti nei lavori svolti durante questi tre anni ed ai Professori che hanno saputo darmi utili consigli nel momento di decisioni importanti. Cito, tra i tanti, il Prof. Guidetti, il Prof. Pessina, il Prof. Fiala, la Prof.ssa Benedetto, il Prof. Olper e la Prof.ssa Casiraghi.

Grazie a tutti i colleghi con i quali ho condiviso questi importanti anni, tra gioie e dolori….e un grazie speciale a Jacopo, amico e collega senza il quale molti dei lavori svolti non avrebbero visto la luce. Grazie anche a Marco, per il supporto tecnico e morale nonché per l'organizzazione delle mangiate al Doge…

Grazie agli amici di Manchester, per aver reso più facile la mia permanenza lontano da casa.

Grazie alle amiche su cui ho sempre potuto contare, Ilar, Bea, Ale.

Grazie alla mia famiglia, ai miei genitori, a mia sorella, a mia nonna, ai miei suoceri e cognati, e un grazie particolare a mio padre per il prezioso aiuto nei momenti di smarrimento.

Grazie a Carlo perché è il mio punto di riferimento, la mia casa, il motivo della mia serenità.

And a special thanks to Adisa, for being such a good supervisor during my visiting period at The University of Manchester, and for having given me the great opportunity of working in her Group again.