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Application and enhancement of Life Cycle Assessment and Water Footprint approaches to agricultural machinery and cultivation

PhD Thesis: Daniela Lovarelli

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List of publications

1) **Lovarelli, D.,** Bacenetti, J., Fiala, M. (2016a). Life cycle inventories of agricultural machinery operations: a new tool. Journal of Agricultural Engineering. XLVII, 40-53.
2) **Lovarelli, D.,** Bacenetti, J., Fiala, M. (2017). Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. Journal of Cleaner Production. 140, 479-491.
3) **Lovarelli, D.,** Bacenetti, J. (2017). Seedbed preparation for arable crops: environmental impact of alternative mechanical solutions. Soil Tillage & Research. 174, 156-168.
4) Bacenetti, J., **Lovarelli, D.,** Fiala, M. (2016). Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation. European Journal of Agronomy. 79, 107-118.
5) **Lovarelli, D.,** Fiala, M., Larsson, G. (2018). Fuel and engine emissions during on-field tractor activity: a possible improving strategy for the environmental load of agricultural mechanisation. *Under review.*
6) **Lovarelli, D.,** Bacenetti, J., Fiala, M. (2016b). Water Footprint of crop productions: A review. Science of the Total Environment. 548-549, 236–251.
7) **Lovarelli, D.,** Ingrao, C., Fiala, M., Bacenetti, J. (2016c). Beyond the Water Footprint: a new framework proposal to assess freshwater environmental impact and consumption. Journal of Cleaner Production. 1-11.

The contribution of Daniela Lovarelli to the papers included in this PhD Thesis was as follows:

1) Planned the work with the co-authors, wrote the paper with input from other co-authors and performed with some assistance the short case study.
2) Planned the work with the co-authors. Performed with some assistance the case study and data analysis. Wrote the paper with input from other co-authors.
3) Planned the work with the co-author. Wrote the paper, realised the case study and data analysis and interpretation.
4) Planned the work with the co-authors. Participated in the realisation of the case study and data analysis. Wrote the paper with other co-authors.
5) Planned the work with the co-authors. Wrote the paper, realised the case study and data analysis and interpretation with collaboration of co-authors.
6) Planned the work with the co-authors. Collected information from literature and wrote the paper and the data interpretation.
7) Planned the work with the co-authors. Deepened the analysis of the case study of Paper 4, performed data analysis, interpretation and wrote the paper with collaboration of co-authors.

1. Introduction

It is well known that the environmental sustainability and the environment-related concerns are under a particularly important and worldwide growing interest (IPCC, 2006; Notarnicola et al., 2015). Attention to sustainable production systems (Notarnicola et al., 2017) and bio-based economies (Ingrao et al., 2016) and promotion of the life cycle thinking are key priorities for the near future productions. In order to reduce the gap between Earth resources availability and the humankind intervention on resources extraction, transformation and use, a transition to sustainable productions with low production inputs and low pollutant emissions is needed.

On a worldwide scale, all production sectors are responsible for part of the environment pollution and resources depletion. Among them, agriculture heavily contributes to environmental impacts (IPCC, 2006). It is a sector, however, on which efficiency improvements have already been introduced and that can still achieve interesting mitigation progresses due to the intrinsic relationship with the environment. Climate change is the most widely known and studied issue (IPCC, 2006), but there exist several other conditions that cause heavy damage to the environment and that are involved in undesired processes such as eutrophication, acidification, ozone layer depletion, mineral, fossil and renewable resources depletion, water depletion, etc. (ILCD Handbook, 2011).

The Intergovernmental Panel on Climate Change (IPCC) deeply investigated the effects of the increase in anthropogenic greenhouse gases emissions, which have led to an increase of concerns about global warming and environmental issues. In the agricultural sector, the major responsibilities for global warming are attributable to (IPCC, 2006):

- enteric fermentations from animal livestock;
- manure and slurry management;
- mineral fertilisers production and related emissions;
- paddy rice cultivation;
- production inputs for crop cultivation (e.g., fuel, fertilisers and pesticides, machinery).

All of them play a role on the environmental pollution. Emissions of methane (CH_4) are mostly due to animal enteric fermentation and paddy rice cultivation, those of nitrous oxide (N_2O) derive mainly from field processes related to organic and mineral fertilisers management and, lastly, emissions of carbon dioxide (CO_2) are mostly due to the production, transport, use and disposal of inputs such as of the fuel used during agricultural mechanisation processes. As widely known, these gases are among the major causes for global warming as well as for other environmental impacts such as eutrophication, acidification, particulate matter formation etc., and their effect must be evaluated by means of characterisation factors.

With a focus on crop cultivation, both for food and feed purposes, the main responsibilities on the environmental point of view are related to the production, transport and use of inputs (e.g., fertilisers, pesticides, irrigation water, fuel, lubricants and materials of which are made the machines) (Notarnicola et al., 2015; Fusi et al., 2014a). Agricultural machinery operations and the emission of substances (e.g., organic and mineral fertilisers, pesticides, engine exhaust gases) into air, soil and water in fact represent the most relevant processes for environmental impacts related to the on-field cultivation phases (Renzulli et al., 2015; Schmidt Rivera et al., 2017). Moreover, agricultural productions are the key responsible for worldwide freshwater use, causing effects on the environment in terms of water consumption, depletion and pollution (Lovarelli et al., 2016b; Mekonnen and Hoekstra, 2011; Pfister et al., 2009).

These conditions especially describe the Italian productive context, Po Valley in particular, where the agricultural production systems (crop cultivation, livestock, etc.) are intensive. The inputs use is very high (e.g., organic fertilisers spreading due to the intensive livestock, mineral fertilisers and pesticides employed to support crop growth, freshwater use due to the high water availability and complex water network, and fuel used during all field operations), primarily thanks to the local context and intensive production, to the favourable climate and to the frequent presence of double cropping systems. Thus, mechanisation substantially participates in the effects on the environmental sustainability of agricultural productions.

Since agricultural systems are based on natural local conditions and are characterised by complex processes and by a high number of variables and alternative machines, difficulties in data collection have arisen. Consequently, cultivation processes often show modelling simplifications (Lovarelli et al., 2016a) characterised by various levels of accuracy.

In general terms, data to fulfil inventories can be obtained with two main sources that consist of:

- primary data, they are directly measured or collected in the system or study area. Primary data are reliable, geographically and temporally specific, but can be difficult to get, expensive and time consuming. Moreover, their positive effect may be only related to the measurement area or context, making those values unadapt to any other use;

- secondary data, they are obtained from database and literature. They lack in information specific for site and temporal locally performed studies and processes, since they commonly represent average conditions, unless the study area of interest corresponds to that gathered from secondary data. This may cause less reliability in the evaluations, due to modelling simplification. However, they can be identified as not overly time-consuming data to collect.

1.1 Life Cycle Assessment

In this context, a method to uniquely quantify the environmental impact of agricultural productions is fundamental.

The Life Cycle Assessment (LCA) is a holistic method recognised worldwide for quantifying the potential environmental impact of productions and/or services (ISO 14040 series, 2006) analysing the whole life cycle. Given its synergetic possibilities, it has become globally adopted in several production sectors, among which agriculture (Notarnicola et al., 2017).

To carry out an LCA study, four phases must be completed. Among them, the inventory is a very important one, although difficult to fulfil. In this phase, all data regarding flows and masses of inputs and outputs that concern the system must be collected.

During this PhD Thesis, with the application of LCA to agro-food and feed productions, the main problems of this method – already highlighted in other studies (Bacenetti et al., 2015; Lovarelli et al., 2016a, 2017; Sala et al., 2017) - emerged. The collection of reliable inventory data (i.e. primary data or locally valid secondary data) is at the basis of a well-performed study; thus, the more reliable is the inventory, the more reliable are the environmental impact results. However, data collection can be very difficult and for this reason, the use of databases and literature (i.e. secondary data) is widespread. In particular, these last can be even the most adopted solution from practitioners, since they allow avoiding unwanted double counting or lack of information. Additionally, especially when assessing agricultural processes, not always reliable local inventory data and context-adapt emissive factors for impact categories can be obtained (Sala et al., 2017). In more details, for agro-food productions, and, in particular, for field mechanisation processes, data collection is one of the most difficult activities (Lindgren, 2005; Janulevičius et al., 2013). Among the most important issues related to the influence of local data on the environmental assessments there are local pedo-climatic and operating variabilities that define the working context and that involve substantially different environmental impact results (Bacenetti et al., 2017; Lovarelli and Bacenetti, 2017). In particular, inventories play a very important role since every LCA study focuses on a specific system, with a defined system boundary and functional unit. Therefore, results can differ considerably simply improving the efficiency of an operation.

The most common database used in LCA studies is Ecoinvent® (Althaus et al., 2007; Frischknecht et al., 2007; Jungbluth et al., 2007; Weidema et al., 2013), which is a very useful, transparent and promising data source available in the LCA software (i.e. Simapro, Gabi). However, it can report average data not always adequate to describe the studied system (Nemecek and Ledgard, 2016). In fact, database average data are included and information about some operations and/or about new machinery with new standards and technologies is missing, causing unrepresentativeness of some systems (Lovarelli et al., 2017).

It is also important to consider that the availability of innovative technologies allowed for developing interesting solutions for data collection, which can be helpful to improve the reliability of inventory data and their application to exhaustive studies. In more details, it is possible to assemble tools such as CAN-bus (Controller Area Network), GPS (Global Positioning System) and electronic devices on modern tractors that permit to gather a huge amount of primary data in an easier way, directly while working on field (Bacenetti et al., 2017; Marx et al., 2015; Pitla et al., 2016). Although mainly used for several goals in precision agriculture, these instruments are used promisingly to collect information about the engine (engine speed, engine load, torque, fuel consumed, etc.) (Fellmeth, 2013; Lindgren, 2004; Speckmann and Jahns, 1999), about the working conditions (working speed, slipping, etc.) as well as about the geographical position of the tractor (Bacenetti et al., 2017; Pitla et al., 2016).

Finally, the results of LCA studies performed on the same product cannot be compared if the system boundary or the functional unit are not completely the same. The reason is that different assumptions and scopes can determine much dissimilar results. Moreover, if the inventory is not representative of the selected system, the potential environmental impact results can drive to wrong quantifications (i.e. a process results having a higher or lower potential impact than what is effectively responsible for) and even wrong conclusions (i.e. a process results worse than another, although this may be not true). Additionally, it is important to give processes their effective weight in order to avoid studying too much in depth some processes and lose sight of the goal, or vice versa.

1.2 Water Footprint

In relation to agricultural productions, also the role of freshwater-related operations must be evaluated in the context of the environmental sustainability. In fact, freshwater is used in major part for agro-food productions (>70% of worldwide freshwater use) (Antonelli and Greco, 2013). Its use affects freshwater consumption and pollution, mainly because of the release of nutrients and pesticides in the environment and to the energy used for pumping water in systems (Mekonnen and Hoekstra, 2011; Pfister et al., 2009; Lovarelli et al., 2016b).

The considerable role of agriculture on freshwater consumption is partially due to the fact that also precipitation (rainfall, snow) can be included in this evaluation. In fact, although without human roles, if rainfall lacks irrigation water must be applied. Moreover, other field operations affect freshwater use and pollution; for example, while distributing nutrients with mineral and organic fertilisers, nutrients' leaching and run-off can be very important processes of emissions in air, soil and water. This determines a not negligible pollution of the system and causes freshwater eutrophication, acidification as well as ammonia emissions to air (Bacenetti et al., 2016a). In addition, toxicity can also be a problem when active principles are released after pesticides application, both in intensive and extensive production systems. During pesticides spraying, a freshwater volume is also applied, which involves freshwater consumption as well. Usually, the major responsible for freshwater consumption during on-field operations is irrigation, since crops commonly need high freshwater volumes to grow. Instead, during post-harvest operations, processes such as washings and treatments play a role on freshwater use. Generally, freshwater can have (i.e. caught from the natural systems above and below ground through pumping) or not (i.e. precipitation) environmental impacts.

In order to respond to a general need of knowledge about freshwater use worldwide, the Water Footprint (WF) indicator was developed by Hoekstra and Hung (2002) with a volumetric perspective of water consumption. Although this indicator is widely adopted, some methodological simplifications emerged along time, especially when focusing on agricultural productions and, specifically, on freshwater field application and pollution.

Again, similarly to the LCA approach, simplifications arise from the complexity of the agricultural system. To consider the WF of the product and its environmental impact in relation to water consumption and degradation, ISO 14046 (2014) was published to standardise the methodology for WF assessment in compatibility with the LCA product-based approach.

2. Objective

The aims of this PhD Thesis are multiple and follow the identified two main topics.

As concerns the **Life Cycle Assessment (LCA)** the aims are:

- studying the environmental impact of crops production paying attention to local pedo-climatic conditions, operating features, temporal and geographical variability and alternative mechanical solutions that all affect the inventory fulfilment. This permits to quantify the environmental impact of agricultural productions in Po Valley and to identify the process hotspots with specific information related to the productive location;
- understanding how important is the effect of local data on the environmental impact assessment compared with database average data applied in the same contexts. This permits to identify and try to close the gap among different data sources and quantifying correctly the environmental loads;
- improving the methodological framework for reliable modelling and data collection about field operations, in view of efficiency improvements and of the consideration of technological innovations;
- developing a modelling tool that works with local pedo-climatic, temporal, geographical and mechanical variables that mostly affect the systems and that, meanwhile, is robust enough to be trustworthy for local inventories fulfilment. In particular, a tool formerly developed has been improved from its original version and the modelling of fuel consumption and engine exhaust gases emissions has been performed within a specific study during which primary data were collected on field.

As concerns the **Water Footprint (WF)** the aims are:

- analysing the methodological framework of WF, the indicator and its components (i.e. green, blue and grey) to structure a literature review of the studies carried out about agro-food productions, and critically reviewing the Water Footprint approach;

- identifying the methodological uncertainties, drawbacks and/or pitfalls of the methodology and develop and propose an improvement for the WF application to agro-food products. In particular, a framework involving a change in the blue component assessment and the introduction of the Pollution Water Indicator (PWI) is given.

3. Structure of the Thesis

The PhD Thesis is composed of two sections:

- introduction to the topics, goal and scope, methods, results and conclusions of the research project and potential future developments;
- scientific contributions published within the scopes of the Thesis to support the environmental sustainability of agricultural productions.

As shown in **Figure 1**, the following steps were performed.

Firstly, since it emerged that the main drawback of Life Cycle Assessment (LCA) on agro-food productions is the partial lack of inventory data about processes, among which mechanisation ones, and of locally reliable data, a system to improve this point was pursed. In particular, the tool ENVIAM (*ENVironmental Inventory of Agricultural Machinery operations*) was developed and improved from a first release and its detailed methodological framework was described (***Paper 1***). The usefulness of this tool is the support to filling the inventories for agricultural machinery operations considering local variables.

Then, since the benefits of ENVIAM application had to be proved and quantified, case studies were performed. The aim was to identify how relevant is the importance of reliable local inventory data on the quantification of the potential environmental impacts of agricultural cultivation processes by means of LCA. Case studies aimed at studying alternatives for primary soil tillage (***Paper 2***) performed on fields with different soil texture and with different implements, and for the seedbed preparation (***Paper 3***) involving both primary and secondary soil tillage operations as well as minimum tillage alternatives.

Additionally, since an important issue in Po Valley concerns fertilisers application and emissions to air, soil and water, the use and effect of different machinery and mechanisation solutions for spreading organic fertilisers were evaluated and compared on the environmental point of view. In more details, direct injection and surface spreading with different timing in the incorporation were assessed (***Paper 4***); moreover, the study included a comparison among different fertilisers, both organic and mineral, as well as different residue management strategies. Their environmental load was quantified and mitigation strategies were suggested.

Having seen that ENVIAM has a beneficial potentiality on the support to the quantification of environmental loads of agricultural machinery field operations due to the local-specific variables, an additional improvement in the quantification capabilities was desirable. Consequently, fuel consumption and exhaust gases emissions were studied during a field experiment. Direct measurements on field were carried out with CAN-bus (Controller Area Network) and data logger, GPS (Global Positioning System) and engine exhaust gases emission analyser. The measures were performed in Umeå (Sweden) during an internship at the Swedish University of Agricultural Sciences (SLU; Uppsala, Sweden). Data processing aimed at studying the single phases that characterise a field operation (i.e. effective work, turns at the headlands, stops and transport phases from farm to field and vice versa). In parallel with data processing, a reliable engine-specific model was applied and described (***Paper 5***).

Finally, regarding freshwater concerns in the environmental sustainability of agro-food productions, the Water Footprint (WF) indicator was studied.

Firstly, this purpose was reached by completing a literature review (***Paper 6***) for enhancing knowledge about the WF concept and application, about the numerous studies present in literature, and for clarifying why and to what extent several methodological concerns were found in literature.

Secondly, a critical methodological analysis of the WF definition and calculation method was completed. Calculation improvements resulted highly recommended; thus, a framework was proposed. The blue water component was modified in its calculation assessment (i.e. Water Footprint Applied, WFA) and the Pollution Water Indicator (PWI) was developed (***Paper 7***). Both WFA and PWI were applied to a case study close to Paper 4, expanding the concept of nutrients leaching and run-off and of pesticides application in relation with both their environmental impact and their Water Footprint.

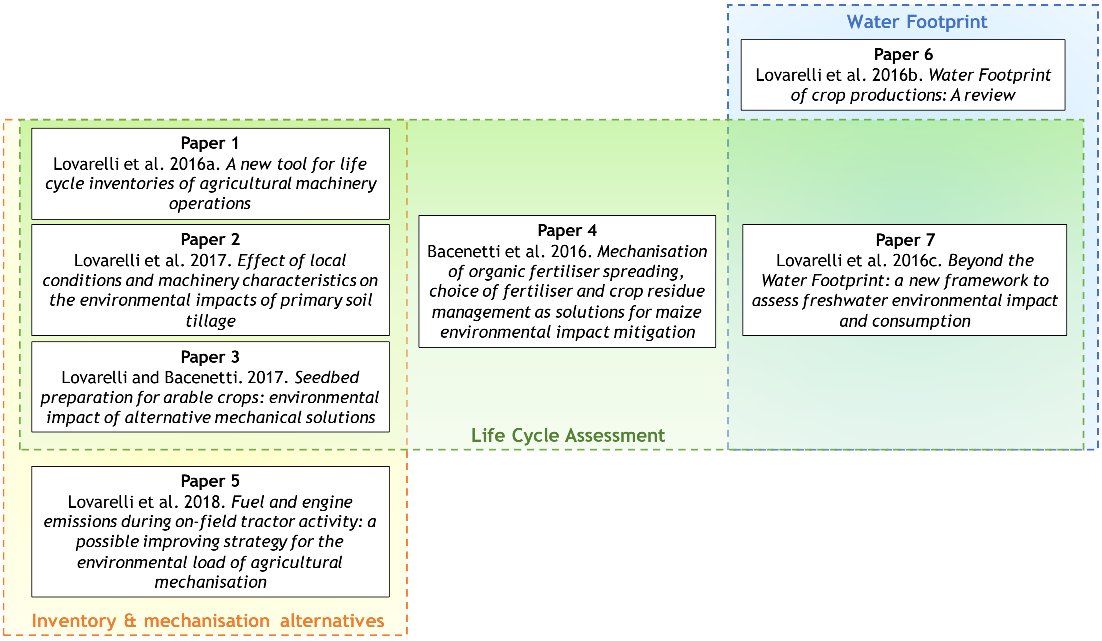


Figure 1. Sequence of scientific contributions.

3.1 Other publications related to the thesis

During the PhD research period, also a number of other related areas of study were investigated. These results were published, but are not widely interconnected with the main aim of the PhD Thesis. However, since they fall within the scopes of mechanisation of agricultural operations in Developing Countries and also provide information on additional concerns of mechanisation, or they are related to Life Cycle Assessment mitigation evaluations (of the milking and anaerobic digestion sector), they are listed below and cited in the PhD Thesis where appropriate:

- Lovarelli, D., Bacenetti, J., Tholley, J., Fiala, M. (2017). Comparison between two rice cultivation practices in Sierra Leone: traditional and alternative methods. Agricultural Mechanization in Asia, Africa and Latin America. *In press.*
- Lovarelli, D., Bacenetti, J. (2017). Bridging the gap between reliable data collection and the environmental impact for mechanised field operations. Biosystems Engineering. 160, 109-123.
- Bacenetti, J., Bava, L., Zucali, M., Lovarelli, D., Sandrucci, A., Tamburini, A., Fiala, M. (2016). Anaerobic digestion and milking frequency as mitigation strategies of the environmental burden in the milk production system. Science of the Total Environment. 539, 450-459.
- Bacenetti, J., Lovarelli, D., Ingrao, C., Tricase, C., Negri, M., Fiala, M. (2015). Assessment of the influence of energy density and feedstock transport distance on the environmental performance of methane from maize silages. Bioresource Technology. 193, 256-265.

4. Background

4.1 Definition of Life Cycle Assessment

The Life Cycle Assessment (LCA) is a holistic approach worldwide adopted for quantifying the potential environmental impact associated with all the stages of products and/or services' life cycle (i.e. raw materials extraction, processing, manufacture, transport and distribution, use, repair and maintenance, and recycling or disposal) (SETAC, Society of Environmental Toxicology and Chemistry).

The product/service is studied from the "cradle" (i.e. raw materials extraction from natural resources), through production, transport and use, to the "grave" (i.e. disposal or recycling). As depicted in **Figure 2** as example, masses used and emitted and flows of energy are quantified both in terms of inputs produced and used as well as of outputs released to the environment. All of them are transformed subsequently in values of potential environmental impact by means of characterisation methods.

LCA has the following pluses:

- analysing processes along the life cycle, considering all inputs and outputs with a comparable unit (Functional Unit);
- quantifying the environmental impacts by means of methodologically standardised characterisation factors and impact categories;
- identifying the production hotspots of productive systems to understand where in the cycle an improvement can be relevant;
- avoiding the displacement of problems from a production phase to another and from an environmental impact improvement to another;
- highlighting the production systems' complexity, even for those systems that may appear simple;
- supporting policy makers and stakeholders and providing a basis for informed decisions.

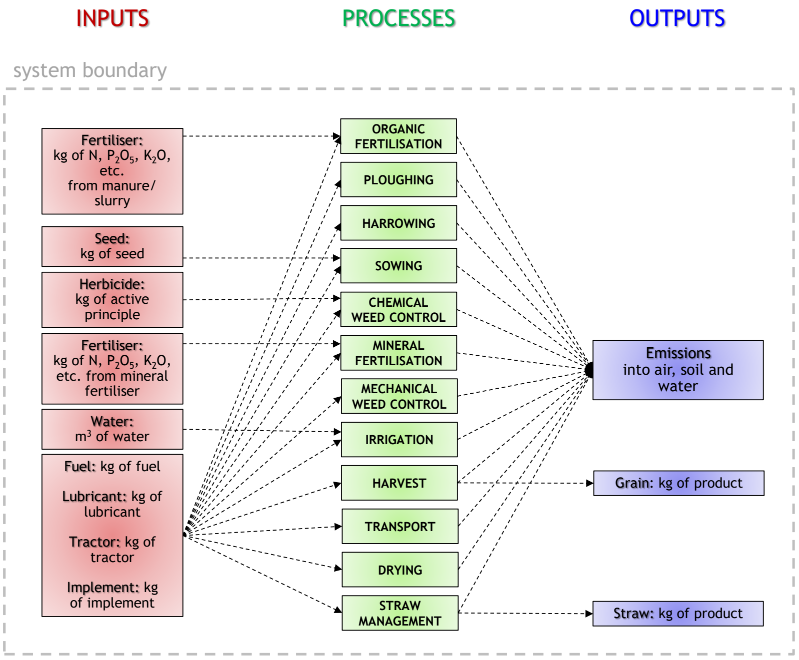


Figure 2. Example of a system boundary for cropping system in which are shown the inventory data.

Given the importance and scientific acknowledgement gained by LCA over the years in several production sectors, among which agriculture, standard rules were developed by the International Standards Organisation (ISO 14040 series, 2006).

4.1.1 Phases of LCA

In ISO 14040 series, four phases were identified for accomplishing LCA:

(i) goal and scope definition:
  a. the product and purpose of the study must be specified,
  b. the system and the system boundary must be defined, which means that the processes included and those excluded from the study are stated, and
  c. a proper Functional Unit (FU) must be selected. The FU is the unit to which are referred all inputs and outputs of the study. The FU is chosen to represent the function of the system; therefore, it must be selected in accordance with the goal and scope;

(ii) Life Cycle Inventory (LCI): the inventory must be filled including all flows/masses of inputs used and outputs produced and released (e.g., amount of fuel consumed, fertiliser spread, and materials depleted);

(iii) Life Cycle Impact Assessment (LCIA): inputs and outputs are converted to environmental impacts by means of classification and characterisation. The potential impacts consist of several environmental impact categories defined at a midpoint level (suggested by ISO), but can also be summarised in impacts on the environment, human health and biodiversity at an endpoint level. To complete this phase, different characterisation methods are available and can be selected (e.g., ILCD, Recipe);

(iv) interpretation:
  a. results are interpreted,
  b. process hotspots are identified, which means that the processes playing the key role on every environmental impact category are recognised, and
  c. mitigation strategies are suggested to reduce the environmental impact of the product and to improve its sustainable production.

4.2 Criticisms about the application of LCA to the agricultural sector

Considering the application of LCA to the agricultural sector some critics have arisen; they mainly refer to:

(i) selection of the proper FU,
(ii) inventory reliability,
(iii) characterisation methods for the environmental impact categories that may result not widely comprehensive and may include uncertainties.

In the following sections, these arguments are explained in more detail. However, it must be stated that LCA is only one of the tools available for environmental assessments, and other tools can be proper for specific studies and be used in combination with LCA for extensive evaluations.

4.2.1 Goal and scope: selection of the Functional Unit

In the first phase of LCA, where the assumptions and declarations are made, the selection of the Functional Unit (FU) plays an essential role on the subsequent phases, and its selection can cause debates.

Usually, the mass of product (1 t) or the worked area (1 ha) are selected as FUs (Renzulli et al., 2015) for agricultural products. However, with the increase of expertise on the topic, authors have started suggesting the use of more adequate and specific FUs (Notarnicola et al., 2017; Sala et al., 2017), such as the nutritional or nutraceutical role of food and feed (crude proteins, energetic content, fats, omega3, etc.). In fact, the FU should be selected at best with the aim of uniquely quantifying the function or the performance of the product and not of describing the general production. However, no comparison can be done among studies characterised by different FU, although they analyse the same system.

4.2.2 Life Cycle Inventory: data collection

The Life Cycle Inventory (LCI) phase can be completed with the help of dedicated databases (e.g., ECOINVENT®, Agri-footprint, Food LCA-DK, EU and Danish Input Output) available in the most used software for LCA studies (e.g., Simapro, Gabi) (Blonk Consultants, 2014; Frishknecht et al., 2007; Nielsen et al., 2003; Weidema et al., 2013), but also by modifying the processes already available in the software.

Differently from industrial processes that are quite standardised and for which databases can be more widely and optimally used, agricultural processes are characterised by intrinsic variability (Notarnicola et al., 2017). In particular, all agriculture-related productions are subject to variability due to pedo-climatic and operative conditions such as: weather and seasonality (e.g., temperature, precipitations, freshwater availability, wind, humidity), presence of weeds, insects and pathogens, natural processes, soil profile (e.g., texture, organic matter, pH) and operating conditions (e.g., field shape, tractor and implement availability and choice, working speed, effective field capacity) as well as inputs application (e.g., rate of organic and mineral fertilisers, pesticides and herbicides).

Given these intrinsic differences, it is very difficult to realise reliable regionalised and well-documented databases adoptable for all of the different pedo-climatic and operating conditions, adapt to different countries, different market machinery and technologies, etc. (Schmidt Rivera et al., 2017). In fact, the average conditions used in databases can be totally misleading when average conditions are unrepresentative of the system. In addition, models are often validated in definite conditions, which means that practitioners should be aware of the model characteristics and of its specific limits and uncertainty. It is highly desirable to use such models with a critical spirit rather than simply using models that operate in widely average conditions while being uncritically conscious of their results.

4.2.3 Life Cycle Impact Assessment: characterisation and impact categories

Another methodological aspect to consider is related to the definition and quantification methods of each environmental impact category, to their uncertainty, and to the possible introduction of other impact categories (Notarnicola et al., 2017; Sala et al., 2017).

Impact categories traditionally rely on non-spatial and steady state environmental models that can be inadequate for agricultural studies in which natural resources, water, land and biodiversity vary with a local extent (Antón et al., 2014). Moreover, model-specific variability and variables included in each impact category and its characterisation may represent a simplification for some authors. However, there are worries for avoiding inserting additional complexity to LCA methodology, since it could undermine its effective added value. The scientific community has achieved convergence among the categories and impact categories' methods are continuously under revision to be updated with the scientific improvements and model robustness.

4.3 Why using LCA for studying agricultural field operations

As mentioned above, LCA has undeniable benefits in quantifying the potential environmental impacts of products, identifying process hotspots and improving their sustainability. Moreover, it is useful for comparing products, marketing purposes and decision-making.

Paying attention to the field mechanisation processes, thanks to LCA is possible to compare (with the same FU) alternative operations or techniques that bring to the same process result and to identify hotspots and best solutions on the environmental perspective. The need of comparing alternative tractors and implements is due to the fact that on the market are present several alternatives, such as different ploughs (mouldboard, slatted and disc) characterised by different working width, working depth and mass which all affect the environmental loads. In particular, (i) implement width, working depth and speed affect the tractor engine power and engine-related variables, the duration of the operation and the distinction of the work phases (e.g., effective work, number and duration of the turns at headlands) and (ii) machinery mass affects the materials depletion (i.e. consumption and wear of materials) along the operation as well as along the life span.

Clearly, also for tractors are available several alternatives (different engine power, mass, gearbox, electronic instrumentation on board, etc.) that must be evaluated with their characteristics. Among others, mechanical features such as power (kW), torque (Nm), engine speed (rad/s) and engine load (%) affect fuel consumption and engine exhaust gases emissions (Jahns et al., 1990; Lindgren and Hansson, 2002) which are fundamental parameters to consider in LCA of agricultural productions. If a tractor is coupled with an implement without considering the optimal coupling and driving, fuel consumption and engine exhaust pollutants increase. Other issues about agricultural machinery (when applicable) concern lubricant consumption and tyres abrasion that both depend on the working time of the operation and on the life span of the tractor (maintenance schedule, work conditions, etc.) and/or implement and they both impact on the environment.

When operations such as fertilisers' distribution or pesticides' spraying are studied, also weather and pedo-climatic conditions must be evaluated because they affect the distribution of products and the emission of related substances. For example, with fertilisers' distribution, together with the environmental load of their production and application, the emissions to air, soil and water occurring during/after the distribution cannot be neglected. Seasonality and weather (mainly temperature, precipitation, wind speed and soil texture) (Brentrup et al., 2000) affect the chemical and physical processes and the consequent leaching and runoff. In addition, considerable importance is related to the incorporation timing of organic fertilisers: in this case emissions are released both to air (e.g., ammonia) and to soil and water (e.g., nitrates) but their contribution depends on the temporal distance between the fertiliser distribution and its incorporation, which can vary between zero (i.e. direct injection of the fertiliser into the soil) and hours or even days (i.e. surface spreading).

The trade-off between emissions to air and to soil and water must be evaluated in detail and with a wide and complete vision, achievable thanks to the LCA approach, in order to avoid optimising the emissions to air (i.e. using techniques that lead to less ammonia emissions) while causing high emissions to soil and water (i.e. using techniques that lead to run-off and leaching), and vice versa (Bacenetti et al., 2016a; Lovarelli et al., 2016c).

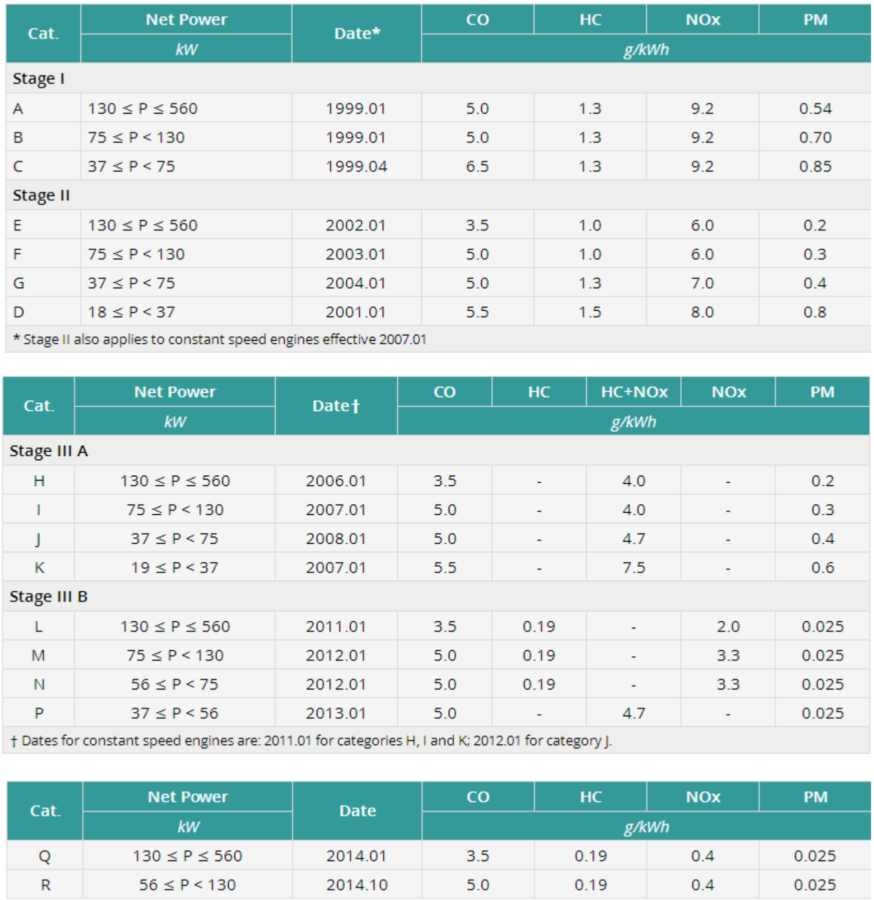
As a proof of the growing role of LCA in system analysis and decision-making (Nemecek et al., 2015; Niero et al, 2015), literature presents a very high number of studies carried out to mitigate the environmental load of agricultural productions. Among them, Bacenetti et al. (2015) and Lovarelli et al. (2016a, 2016b) evaluated the effect of local data compared to the most spread database for LCA studies (ECOINVENT®) in the context of agricultural mechanisation. From these, the positive role of local data availability on the environmental outcomes emerged. Additionally, studies were performed on specific crops (Bacenetti et al., 2015; Fusi et al., 2014a; Negri et al., 2014b) and/or products (Bacenetti et al., 2016b for milk; Fusi et al., 2014b for wine) to quantify the environmental load of their production, from which the responsibility of mechanisation on the environmental point of view widely emerged. Moreover, studies about bioenergy production (e.g., anaerobic digestion) were completed as well with focus on the cropping system phases and on alternative mitigation solutions (Negri et al., 2014a, 2014b; Bacenetti et al., 2014; Lijó et al., 2014a, 2014b; Gonzales-Garcìa et al., 2012).

Nevertheless, very few studies specifically focus on agricultural machinery operations and on their environmental impact, and higher focus should be paid on them. Among the pollutants released during agricultural field operations, the exhaust gases emissions play a very important role, but their quantification is still quite difficult. Especially when considering the big effort undertaken by manufacturers for emissions reduction, the choice of a tractor and the age of its engine can make very important differences on the environmental perspective (Lovarelli and Bacenetti, 2017; Janulevičius et al., 2016, 2017), underlining how relevant it is to have representative data.

4.4 Legislation for exhaust gases emissions of off-road vehicles

In recent times, legislation about exhaust gases emissions of vehicles, trucks, waterway vessels, railway locomotives and off-road vehicles was introduced to reduce air pollution.

Stringent levels of emissions have been defined within European Emissive Stages (or Tiers in US regulation); Stages I-IIIB were specified with the EU Directive 97/68/EC and following amending Directive 2010/26/EU and Directive 2010/22/EU, while the most recent Stage IV-V is normed with the EU Regulation 2016/1628. The restricted pollutants are carbon monoxide (CO), hydrocarbons (HC), nitrogen oxides (NO_X) and particulate matter (PM). **Figure 3** reports the emissive limits (g/kWh) for off-road vehicles of every normed engine exhaust gas, which depend on the Emissive Stage, on the engine power class and year of construction of the engine.



Cat.	Net Power	Date*	CO	HC	NOx	PM
	kW		g/kWh			
Stage I						
A	130 ≤ P ≤ 560	1999.01	5.0	1.3	9.2	0.54
B	75 ≤ P < 130	1999.01	5.0	1.3	9.2	0.70
C	37 ≤ P < 75	1999.04	6.5	1.3	9.2	0.85
Stage II						
E	130 ≤ P ≤ 560	2002.01	3.5	1.0	6.0	0.2
F	75 ≤ P < 130	2003.01	5.0	1.0	6.0	0.3
G	37 ≤ P < 75	2004.01	5.0	1.3	7.0	0.4
D	18 ≤ P < 37	2001.01	5.5	1.5	8.0	0.8

* Stage II also applies to constant speed engines effective 2007.01

Cat.	Net Power	Date†	CO	HC	HC+NOx	NOx	PM
	kW		g/kWh				
Stage III A							
H	130 ≤ P ≤ 560	2006.01	3.5	-	4.0	-	0.2
I	75 ≤ P < 130	2007.01	5.0	-	4.0	-	0.3
J	37 ≤ P < 75	2008.01	5.0	-	4.7	-	0.4
K	19 ≤ P < 37	2007.01	5.5	-	7.5	-	0.6
Stage III B							
L	130 ≤ P ≤ 560	2011.01	3.5	0.19	-	2.0	0.025
M	75 ≤ P < 130	2012.01	5.0	0.19	-	3.3	0.025
N	56 ≤ P < 75	2012.01	5.0	0.19	-	3.3	0.025
P	37 ≤ P < 56	2013.01	5.0	-	4.7	-	0.025

† Dates for constant speed engines are: 2011.01 for categories H, I and K; 2012.01 for category J.

Cat.	Net Power	Date	CO	HC	NOx	PM
	kW		g/kWh			
Q	130 ≤ P ≤ 560	2014.01	3.5	0.19	0.4	0.025
R	56 ≤ P < 130	2014.10	5.0	0.19	0.4	0.025

Figure 3. Emissive limits of off-road vehicles from Stage I-IIIB. [Source: dieselnet.com]

Internationally standardised test cycles are completed in accordance with ISO 8178-C1 for exhaust emission measurement for non-road engine applications. ISO 8178-C1 is adopted for emission certification and approval testing in United States, European Union and Japan for off-road vehicles; considering C1 tests, it foresees the test measurement at different standard torque (10%, 25%, 50%, 75% and 100% of rated torque) and engine speed (rated, intermediate and idle speed) made at bench. Measuring emissions with bench test means that strict transient conditions and field variability are not effectively evaluated, thus it can happen that not always the emitted gases respect the emissive limits.

In the context of LCA and, more in general, of environmental attention to agricultural activities, the possibility of collecting data about engine exhaust gases emissions becomes fundamental for a correct quantification of the environmental load of different mechanisation solutions (i.e. old vs new tractor engines). However, this step is still quite difficult to reach and causes a lower degree in the reliability of databases, although the difference among the emission Stages in terms of emitted gases is huge (**Figure 4**). In particular, in the last 20 years, the emission restrictions have become more and more stringent and the reduction in allowed emissions between the most recent Stage IIIB-IV and Stage I stands in a range of about 100 times less emissions for all exhaust gases.

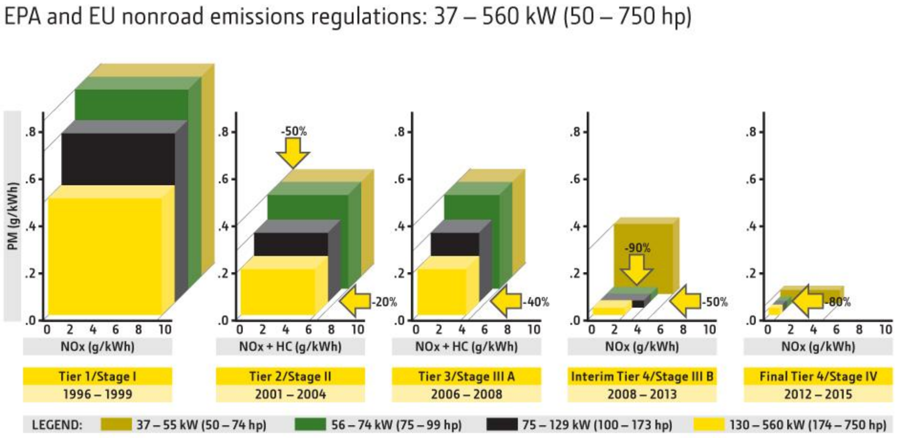


Figure 4. Comparing engine emissions for off-road machinery in the different emissive Stages normed by the EU regulations. [Source: John Deere www.deere.com]

Choosing a tractor that belongs to a different Stage respect to reality brings to consistent and considerable under- or over- estimations of engine emissions and of the subsequent environmental impacts.

4.5 Use and scope of ENVIAM

Having seen the key issues about (i) the inventory fulfilment for assessing the environmental loads of agricultural productions, (ii) the complexity of these production systems and of mechanisation, and (iii) the general global attention on environmental sustainability concerns, ENVIAM (ENVironmental Inventory of Agricultural Machinery operations) is a tool that was built at the Department of Agricultural and Environmental Sciences (DiSAA) of the University of Milan to improve the inventories' applicability and reliability for agricultural machinery operations (Lovarelli et al., 2016a).

Its role within LCA is clear: ENVIAM is used to perform calculations related to mechanisation of agricultural field operations taking into account local variables and characteristics; its outcomes represent the LCI of agricultural machinery operations that bring, subsequently, to the calculation of the environmental impact with LCA software. In **Figure 5** is schematically reported the logical path and functioning of ENVIAM.

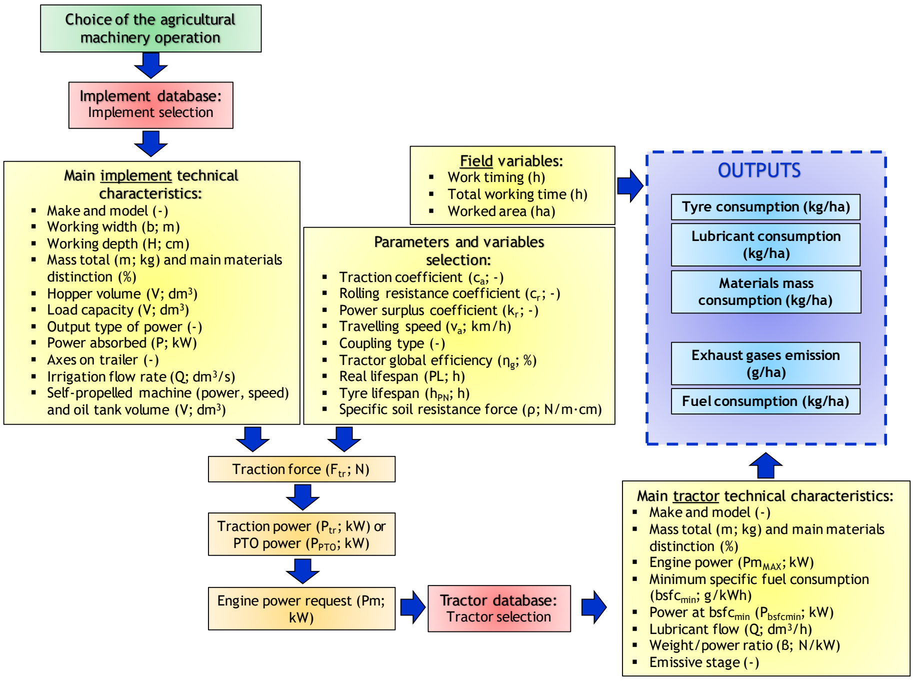


Figure 5. Schematic representation of the functioning of ENVIAM.

One of the most interesting prospects of ENVIAM is that the practitioner can use either the tractor present in the database that best responds to the coupling calculations (i.e. the most efficient choice in terms of inputs use and outputs release) or the tractor that he wants to study. This permits to quantify alternative conditions and, consequently, the effect on the environment of the operative and managerial choices of the farmer.

Moreover, another interesting point is that fuel consumption (kg) and exhaust gases emissions (g) are quantified per working timing, which is characterised by specific engine features. This permits to understand what are the work phases with a higher influence on the inventory data. Therefore, the operation is considered as the total working time gathered from the sum of each working timing, as conceptually shown in **Figure 6**.

Figure 6. Total working time split in single working timings. In general terms, the dimension of the single circles is related to the relevance of those working timings along the whole operation.

In accordance with Reboul (1964), these working timings (hours) are: (i) effective work, (ii) turns at headlands, (iii) transport field-farm and vice versa, (iv) maintenance on farm, (v) preparation of machinery and operator on farm, (vi) preparation of the working layout on field, (vii) refilling/emptying of the fuel tank and/or of the hopper.

ENVIAM represents the first attempt of studying field operations with an environmentally sustainable perspective by considering:

- most options available on the market for tractors and implements, making available an up-to-date database of agricultural machinery;
- the variability of local conditions in terms of working and pedo-climatic features (e.g., characteristics of the specific tractors and implements used, soil texture, field shape, etc.);
- the possibility of studying operations split in the different working states that compose the operation, identifying those working phases that need efficiency improvement or better driving practices (e.g., "gear-up, throttle-down" Grisso et al., 2014);
- the possibility of quantifying the variables that affect fuel consumption and engine emissions in defined work conditions and in each working state, which permits to improve the reliability of the LCI.

4.6 Technological instrumentation for improving the efficiency of mechanisation and enhancing LCA studies

In recent time, the improvement of work and inputs efficiency and the introduction of innovative technologies in the agricultural sector allowed for improving and simplifying the data collection and monitoring on field during agricultural machinery operations. In particular, the introduction of technologies such as GPS (Global Positioning System), CAN-bus (Controller Area Network), electronic devices and gases emission analysers has permitted to study with precision the engine behaviour and the effective features characterising the work on field.

Being the agricultural sector a complex system, characterised by interconnections and variables, the possibility of introducing technological innovations that helps understand and describe this complexity is positive. Optimising inputs use and application is a desirable goal on several points of view: agronomic, economic and environmental. Therefore, precision agriculture in general, but also automation, sensors and devices are gaining wide interest. The applications for advanced technologies are undoubtedly high and, in this PhD Thesis, they focus on the benefits of adopting technological innovations for mechanisation processes and data processing for inventory filling within LCA scopes (Lovarelli and Bacenetti, 2017).

From literature has emerged that the technological instrumentation is gaining importance for several applications. Suprem et al. (2013) reviewed the possible equipment to perform studies in the agro-food sector, showing the important benefits in introducing technological systems, among which CAN-bus. In Lindgren (2004, 2005) is reported a case study in which CAN-bus and exhaust gas analyser were used during several field machinery operations to study fuel consumption and engine emissions while considering the role of transient conditions. Similarly, Janulevičius et al. (2013) studied the effect on field operations of fuel and engine emissions, giving a very interesting result on the eight evaluated tractor models and they further improved their study focusing on the not-to-exceed zones (Janulevičius et al., 2016, 2017). Perozzi et al. (2016) studied the duration of idling conditions along the tractor life span by collecting CAN-bus data. Marx et al. (2015) compared the CAN-bus data collection with the system used by Nebraska Tractor Test Laboratory (NTTL) stating that the difference error for the measurement of fuel consumption between the two methods was quite restrained (6.22%).

This massive amount of data about tractor features, tractor engine and geographical position is very helpful in multiple contexts and permits to achieve several goals, both for practical concerns (maintenance, damages identification, etc.) (Bietresato et al. 2015) and for research-inputs evaluation assessments (Larsson and Hansson, 2011; Lindgren and Hansson, 2004; Mantoam et al., 2016). In particular, engine specific variables can be mapped on field and it can be recognised on each part of the field what is the relevance of the variable (e.g., torque, engine speed) as well as of the fuel consumed and of the exhaust gas emitted to air. For example, Pitla et al. (2016) carried out a study in which the operation was split in different working phases on U.S. fields and the engine variables were evaluated specifically in each phase (i.e. effective work, turns at headlands, etc.). Similarly, Lacour et al. (2014) studied different work phases (i.e. transfers, effective work, turns and stops and goes) to build a model to convert bench tests information in indicators for effective efficiency indicators.

With CAN-bus, engine data and interactions with other electronic components on tractors permit to evaluate instant by instant the changes in variables, and in which working conditions positive or negative effects can be observed. For instance, driving skills and engine power affect fuel consumption, and improving knowledge about them and the related variables permits to increase the operation efficiency and to optimise the fuel use. Linked to fuel, a similar statement can be drawn for the emission of exhaust gases. In particular, variables such as engine speed (rpm), engine torque (Nm), oxygen concentration in the exhaust pipe (%) and exhaust gases temperature (°C) lead to understanding the relationship with emissions and, consequently, to reduce the frequency of conditions that cause the heaviest pollutant emissions. Nonetheless, a solution is not easy, since improving some conditions does not mean improving the exhaust gas concentration of all of the emitted gases: again, trade-offs are needed (Janulevičius et al., 2013; Lindgren, 2004).

Beside the advantages and strengths of instrumentation, there are also limits. The most evident is that data must be collected in the different working conditions and for every tractor engine. In fact, the behaviour of an engine is specific, which entails the necessity of measuring physical quantities per engine. Therefore, the major drawbacks of primary data must (still) be counted: measurements are valid almost only in the specific measurement context and are time consuming.

Still, these measurements can become widely applicable in the near future, first for manufacturers who can adopt electronic devices as a source for testing and setting their machinery and second for users (farmers) to have information about the main settings and possibilities of their machinery, as well as for researchers. Then, the possibility of collecting data during the work activity will permit to develop databases more and more sophisticated and with wide applications.

4.7 Definition of Water Footprint

The Water Footprint (WF) is an indicator of virtual freshwater volume content of products, services and/or communities. In accordance with Pfister and Boulay (2017), this concept was developed by Hoekstra and Hung (2002) and further studied and improved within the Water Footprint Network (WFN) organisation.

However, the freshwater issue is important under several points of view; thus, also the LCA community got interested in it, and introduced a methodological framework able to work with the life cycle concept.

4.7.1 Water Footprint in the WFN approach

WFN defines specific guidelines (Hoekstra, 2010) for quantifying in volumetric terms (m^3) the freshwater necessary to produce or, in other terms, embedded in goods and services that are consumed by individuals or communities. The WF is commonly referred to a unit of product (1 t) or of area (1 ha) and can also be referred to a nation or geographical area using a temporal reference (1 year).

As also illustrated in **Figure 7**, WF in made of three components:

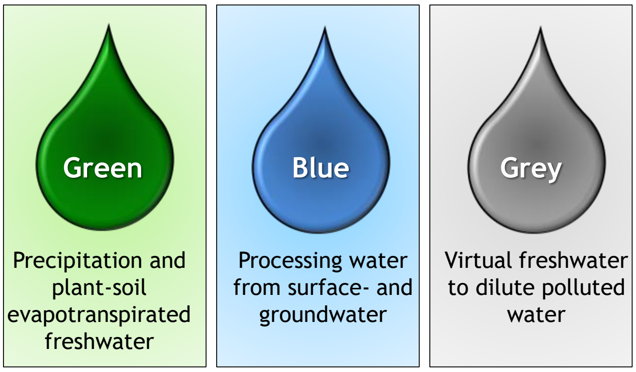


Figure 7. Components of Water Footprint in accordance with Water Footprint Network.

(i) green water: volume of freshwater evapotranspirated from the soil and plant system, it does not have economic costs since is related to the precipitation (e.g., rainfall, snow) got during the cropping season;

(ii) blue water: volume of freshwater pumped in the system with a human intervention. It derives from groundwater (e.g., aquifers) as well as from surface water (lakes, rivers, etc.) and represents the freshwater volume introduced in the system through irrigation, washings, processing water, etc. Therefore, it has both economic and environmental costs;

(iii) grey water: virtual water volume necessary to assimilate the pollutants emitted to water and used during the production. It depends on the normative allowable concentration of the pollutant, on the natural concentration of the watershed and on the amounts released in the productive processes.

4.7.2 Water Footprint in the LCA approach

Meanwhile that WFN studied and introduced methodological changes to the original WF concept, the LCA community identified "Water Use" as a very important variable for environmental impact assessments. This interest brought to the introduction of WF with an LCA-perspective in accordance with the very recent international standard ISO 14046 (2014); this, however, covers different water accounting concepts but used to keep the same name, which has caused lots of debates (Pfister and Boulay, 2017).

ISO 14046 (2014) was introduced for having a standardisation of the calculation method and a common ground with the LCA approach. Accordingly, the WF (ISO 14046 approach) is assessed with an environmental impact perspective following the same four phases adopted in LCA: (i) goal and scope definition, (ii) inventory analysis, (iii) impact assessment, and (iv) interpretation. The aim in this case is the quantification of the potential environmental impact related to water consumption and degradation.

Methods to identify the adequate midpoint impact category for freshwater quantification have been developed and certainly differ from the WF calculation method based on the WFN perspective (Boulay et al., 2013; Lovarelli et al., 2016b). To evaluate water pollution and the effective water availability of a geographical area was introduced, among others, the Water Stress Index (WSI) (Pfister et al., 2009) to consider the water stress of a geographical area. In addition, in 2007 the Water Use in Life Cycle Assessment (WULCA) was founded under initiative of the UNEP/SETAC; WULCA is a Life Cycle Initiative project for assessing use and depletion of water resources within the LCA framework. WULCA developed the Available WAter REmaining (AWARE) method, which is the current recommended framework for WF assessment with LCA perspective (Boulay et al., 2015) and is introduced in LCA software (http://www.wulca-waterlca.org/index.html).

4.7.3 WULCA scope and AWARE method

The WULCA working group's overall goal was to provide practitioners with a consensual and consistent framework to assess, compare and disclose the environmental performance of products and operations regarding freshwater use.

In 2013, to harmonise and build consensus for "Water Use" impact category AWARE was introduced with the aim of understanding *the potential to deprive freshwater users (humans or ecosystems) from freshwater by consuming freshwater in the region*. AWARE is the currently recommended method to assess the impact of water consumption in LCA (Eq. 1) and is defined as *"a water use midpoint indicator that represents the relative Available WAter REmaining per area in a watershed after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived"* (http://www.wulca-waterlca.org/aware.html).

$$AWARE = \frac{1}{AMD} \tag{1}$$

where:

AMD = Water Availability – Water Demand ($m^3/m^2 \cdot$ month), corrected with the AMD world average (0.0136 $m^3/m^2 \cdot$ month).

AWARE is calculated as the available water minus the water demand of humans and aquatic ecosystems, normalised with a global average and inverted. The result is the relative surface-time equivalent value to generate unused water in a region respect to the average volume consumed in the world (where the world average is a consumption-weighted average). The indicator ranges from 0.1 to 100, with a value of 1 corresponding to the world average and, for example, a value of 10 representing a region where there is 10 times less available water remaining per area than the world average.

4.8 Criticisms about the application of WF to the agricultural sector

During the last decade, several studies have been performed globally about Water Footprint (WFN approach) of agricultural productions.

However, except for green water, where the quantification is made following the evapotranspiration model by FAO (Allen, 1998) and the calculation is almost straightforward, the other two components (blue and grey) have been criticised on several points of view and from several authors. Mostly, the reasons are related to the lack of numerous variables (e.g., water scarcity, water stress, geographical local conditions, evaluation of only the pollutant that requires the highest volume for freshwater quality restoration, etc.) (Ridoutt and Pfister, 2010; Lovarelli et al., 2016c).

In more detail, in 2015, Garcìa-Morillo et al. (2015) introduced the blue Water Footprint Applied (WFAblue) concept with regard to the accounting of irrigation within the blue water assessment. A similar critique (and subsequent alternative calculation method) was found in several scientific contributions (Cao et al., 2015; Lovarelli et al., 2016c; Scarpare et al., 2016; Yoo et al., 2013). Additionally, critiques on the grey WF (WFgrey) assessment emerged as well, and often the grey component was not accounted for in WF studies (Lovarelli et al., 2016b) due to lack of information or methodological uncertainty.

4.8.1 Blue water

Blue water in WFN is quantified considering the evapotranspirated water that is embedded in the product, or, in other words, the effectively used water during cropping/processing that does not go back to the water system. It considers groundwater and surface water pumped in the system, but not the irrigation technology and its efficiency (García-Morillo et al., 2015; Lovarelli et al., 2016c).

However, for what regards the agricultural field cultivation phase, it is unacceptable to evaluate only the irrigation water absorbed by the crop because, according to the irrigation method, the effective irrigation volume to apply can differ significantly. This surely has an impact on several points of view. Firstly, the effective needed volume may not be locally available (due to the irrigation network, to irrigation availabilities, water stress and scarcity, etc.) when the crop specifically demands for it. Secondly, the absorption of water from water rich areas can have a different importance respect to water scarce areas, and therefore, the related WF of the blue component (WFblue) cannot be compared between rich and scarce areas. In addition, it cannot be considered sustainable for the near and far future, even if it shows the same value in different areas.

4.8.2 Grey water

Grey water considers the virtual water volume necessary to restore water quality after it has been polluted during the cultivation/processing. It evaluates the maximum allowable concentration of pollutants in water respect to the natural concentration of the watershed, but only quantifies the dilution volume as function of the pollutant demanding the highest dilution volume. Thus, the pollutants present in freshwater that require a lower dilution volume respect to the highest one are totally disregarded for the WFgrey.

Nevertheless, along field cultivation phases, the pollutants and nutrients spread on field are several and their presence and effect may stockpile along time. Consequently, it is not only the substance that requires the highest volume to restore water quality that should be considered. In agricultural productions, in fact, the distribution of fertilisers involves that several nutrients are applied. According to the spreading technique (e.g., surface spreading or direct injection), the leaching and runoff to soil and water and/or the emissions to air of the nutrients vary and have a different impact on the environment (Bacenetti et al., 2016a).

Although WFgrey is evaluated in volumetric terms and with no reference to environmental impacts, it still includes a conceptual inaccuracy because it considers a virtual volume for quality restoration, when, instead, the quality is effectively damaged (Pfister and Boulay, 2017). This means it is not matter of volumes. Moreover, the sum of green, blue and grey water involves summing volumes with different environmental roles, which can be misunderstood or misinterpreted.

Finally, even if the WF has resulted a very interesting indicator, its use lacks information and important geographical and physical issues on the scientific background because, differently from other resources, freshwater is a local one and local information and spatial identification are essential.

4.8.3 Transformation of the critics as basis for LCA approach

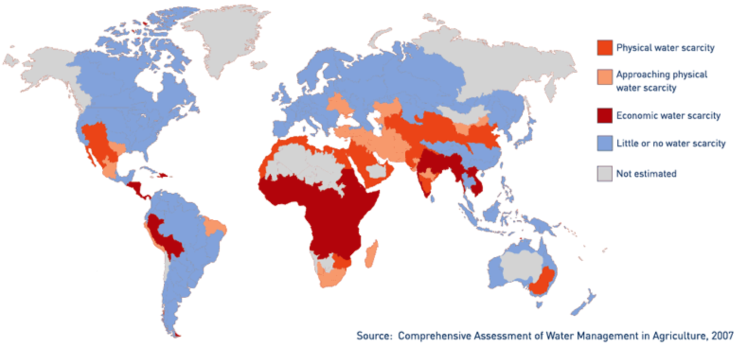
All aforementioned environmental issues had to be considered when quantifying the WF of agricultural products, which motivates why an LCA approach has been searched for WF assessments. Therefore:

- methodological changes to WF were proposed on the basis of the WFN approach;
- the interest in water assessment for the identification of its potential environmental impact increased: ISO 14046 born to clarify the different methodological frameworks and to organise the calculation method on an LCA perspective (with the impact category "Water Depletion" quantified in terms of H_2Oeq). Other studies were also performed until the introduction of "Water Use" impact category defined with AWARE.

4.9 Why using WF for studying agricultural products

Although agriculture plays an essential role for humans and the environment, it is responsible for environmental impacts and for a considerable share of worldwide freshwater consumption and depletion (Mekonnen and Hoekstra, 2011).

In addition, climate change is bringing to changes in worldwide agricultural production habits, due to the increasing water scarcity (**Figure 8**) and extreme precipitation events, and agriculture must adapt.



Source: Comprehensive Assessment of Water Management in Agriculture, 2007

Figure 8. Areas of physical and economic water scarcity. [Source: Comprehensive Assessment of Water Management in Agriculture 2007]

In this context, studying the freshwater volume content of productions is very helpful for becoming aware of water consumption, both in direct and indirect terms. This is especially valid for agricultural products, for which commonly, humans are not at all conscious of their role. Since agriculture has the highest responsibility for freshwater consumption (>90%; Mekonnen, 2011), evaluating the WF of agro-food productions and splitting it in three components with different origin and effect can make clearer to consumers to what are due water consumptive processes and how relevant is the human intervention on them. However, the numerous simplifications bring to important incorrect evaluations on the scientific perspective.

If properly quantified, similarly to LCA, WF can both help become aware of water resource importance and limits and help decision makers develop adequate and consistent policies. Being freshwater a local resource affected by several events (e.g., rainfall, irrigation), it is very complicated to study reaching a good level of approval in the scientific community. Therefore, it has been extensively studied and methodologically modified.

Finally, as already mentioned, the scope of WF within the WFN framework is different from the one with the LCA perspective and the two indicators cannot be compared, but coexist.

5. Materials and methods

This chapter is divided in the following sections:

(i) the methodology and methods adopted for LCA studies and for the inventory fulfilment completed with ENVIAM;

(ii) the materials and methods for primary data collection and processing during field experiments performed at SLU in Sweden;

(iii) the methodology and methods adopted for WF studies.

5.1 LCA and ENVIAM studies

5.1.1 Studies set-up

The lack of local inventory data for agro-food field operations and the uncritical use of Ecoinvent database represent the reasons why ENVIAM was developed and gradually improved; its methodological framework and functioning is described in *Paper 1*[1].

In more detail, ENVIAM is a Microsoft Office Excel file composed of different worksheets in which mechanical information from literature and from the market is retrieved and used for inventory data calculations. ENVIAM is made of two databases listing both tractors and implements that are adoptable for several machinery operations, with focus on the crop production chain. The practitioner can select the operation to carry out, the tractor and implement to use, and the variables that affect the calculations. Among them there are: soil texture, typology of coupling between tractor and implement, Emissive Stage to which the engine belongs, specific soil resistance force, etc.

Additionally, the practitioner must introduce the working time (total and split in working timings) and the worked area. Once these variables are identified, calculations are performed to quantify the engine power request and the optimal rated power for the

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[1] Lovarelli, D., Bacenetti, J., Fiala, M. (2016a). Life cycle inventories of agricultural machinery operations: a new tool. Journal of Agricultural Engineering. XLVII, 40-53.

tractor; afterwards, fuel consumption (Lazzari and Mazzetto, 2005) and exhaust gases emissions (Schäffeler and Keller, 2008), and material and lubricant consumption are quantified.

In the study, beside the methodological description of ENVIAM, a concluding comparison with the Ecoinvent database is made to demonstrate how average inventory data uncritically adopted can be misleading in different working conditions.

Applying ENVIAM, alternative implements and different working conditions (e.g., different soil texture and implement width) were studied considering primary soil tillage (*Paper 2*[2]) and the whole seedbed preparation (i.e. primary and secondary soil tillage operations) also by adding minimum tillage options (*Paper 3*[3]).

In more detail, the assessment about primary soil tillage aimed at studying the effect of ploughing on the environmental point of view by considering the local variability. Therefore, the comparison was made:

- with alternative implements (i.e. mouldboard, slatted and disc plough);
- with a wider plough (i.e. 3 furrows mouldboard plough compared with 5 furrows mouldboard plough);
- on different soil texture categories (i.e. sandy, medium textured and clayey soils) comparing almost all the ploughs alternatives.

Regarding the study on seedbed preparation, the comparison was made considering different sequences of field operations by studying:

- the most common adopted implements in Italy for primary and secondary soil tillage in the crop cultivation system;
- the 3 different soil texture categories (i.e. sandy, medium textured and clayey soils).

All implements were evaluated to complete a sequence of primary and secondary tillage for the seedbed preparation and 13 alternative sequences were built and, when applicable, used on the different soil textures.

In both studies, the operation completed had to bring to the same result on the soil to have the same starting condition (in terms of dimension of soil particles) for the subsequent operations of harrowing or sowing, not included in the system boundary. Still, the Functional Unit (FU) was 1 ha and the system boundary included inputs (energy and

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[2] Lovarelli, D., Bacenetti, J., Fiala, M. (2017). Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. Journal of Cleaner Production. 140, 479-491.

[3] Lovarelli, D., Bacenetti, J. (2017). Seedbed preparation for arable crops: environmental impact of alternative mechanical solutions. Soil Tillage & Research. 174, 156-168.

materials flows) and outputs (emissions to air, soil and water) with an attributional approach and a cradle-to-farm gate perspective. The major interest was paid to agricultural machinery operations; therefore, no crop was considered and the system boundary stopped at ploughing (*Paper 2*) and at harrowing (*Paper 3*).

The environmental outcomes were quantified by means of LCA, modifying the inventory data already available in the processes on Simapro v. 8.0.3.

For what concerns *Paper 4*[4], instead, the aim was to study with the LCA approach the different spreading techniques that can be adopted by farmers, paying attention to the emissions of fertilisers into air, soil and water. The system boundary included the complete cultivation of maize grain, from ploughing to grain drying and straw management. The selected FU was 1 ha and the study had an attributional approach and a cradle-to-farm gate perspective. In the inventory, both collected primary data (interviews with farmers and experts) and secondary data evaluated locally with ENVIAM and with the Ecoinvent database (for background processes) were adopted.

The alternative techniques compared were:

- direct soil injection, where the organic fertiliser is directly injected into the soil. This technique is widely recommended to reduce ammonia emissions to air;
- surface spreading, where the organic fertiliser is spread using a traditional spreader. In this case, ammonia emissions to air are much more relevant and deeply impact on the environment; however, in this case the promptness of the incorporation of the organic fertiliser into the soil is very important on the emissions. Thus, two scenarios were studied: incorporation made within 2 hours after spreading and incorporation after more than 72 hours.

In order to evaluate the fertiliser emissions into the environment, other scenarios were introduced without varying the spreading technique. In more details, considering surface spreading with incorporation after more than 72 hours as baseline scenario, the following cases were studied:

- maize straw collection instead of leaving it on field;
- application of digestate from anaerobic digestion plant instead of pig slurry;
- application of mineral fertilisers instead of organic fertilisers (one scenario with urea and triple superphosphate and one scenario with calcium ammonium nitrate and triple superphosphate).

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[4] Bacenetti, J., Lovarelli, D., Fiala, M. (2016). Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation. European Journal of Agronomy. 79, 107-118.

The emission of fertilisers into air, water and soil was performed with the model by Brentrup (2000) that evaluates also the local pedo-climatic features (e.g., soil texture, slope, wind speed, precipitation).

As regard to the impact assessment, the selected characterisation method for the impact categories was the ILCD (International Reference Life Cycle Data System) (*Paper 2*, *Paper 4*, *Paper 7*); in *Paper 3*, instead, the Recipe method was used. Both methods are the most used for agricultural studies.

In all studies, the contribution analysis was performed with the aim of identifying the processes that played the role of hotspot on the analysed environmental impact categories. Additionally, considerations about the identified solutions and the potential mitigation strategies were reported.

5.1.2 Models adopted in ENVIAM for fuel consumption and engine exhaust emissions

In ENVIAM, fuel consumption was modelled according to Lazzari and Mazzetto (2005). In particular:

$$\mathrm{bsfc}_i = \mathrm{bsfc}_{\min} \cdot \left[2 - \left(2 - \frac{L}{L_0}\right) \cdot \left(\frac{L}{L_0}\right)\right]^2 \quad (2)$$

and

$$FC_i = bsfc_i \cdot Pm_{MAX} \cdot L_i \cdot T_i \quad (3)$$

$$FC = \sum FC_i \quad (4)$$

where:

$bsfc_i$ = brake specific fuel consumption in the *i*-working timing (g/kWh);

$bsfc_{min}$ = minimum value of brake specific fuel consumption (g/kWh);

L_i = engine load in the *i*-working timing (%);

L_0 = engine load that corresponds to the value of $BSFC_{min}$ (%);

FC = fuel consumption (kg);

FC_i = fuel consumption in the *i*-working timing (kg);

Pm_i = engine power in the *i*-working timing (kW);

T_i = *i*-working timing (h).

For engine exhaust gases emissions, the equation by Schäffeler and Keller (2008) was used, since it allowed taking into account the EU regulation with different Emissive Stages. In particular:

$$EM_i = \sum EM_{SP} \cdot Pm_{MAX} \cdot L_i \cdot T_i \cdot CF_1 \cdot CF_2 \cdot CF_3 \quad (5)$$

and

$$EM = \sum EM_i \quad (6)$$

where:

EM_i = emission of exhaust gas in the *i*-working timing (g);

EM_{SP} = specific limit of each exhaust gas (g/kWh) (EU Directive);

CF_1 = correction factor for deviation of effective engine load from the standard load on which the emission factor is based (48%);

CF_2 = correction factor for tractor dynamic use;

CF_3 = correction factor for wear and tear.

Equation (5) was used for quantifying carbon monoxide (CO), nitrogen oxides (NO_X), hydrocarbons (HC) and particulate matter (PM), while for carbon dioxide (CO_2) the factor 3.150 g_{CO2}/g_{FUEL} was employed (Schäffeler and Keller; 2008).

5.2 Data collection during field experiments

With regard to the primary data collection on mechanised field operations assessed and processed in *Paper 5*[5], the field experiments were planned during a visiting period in collaboration with the Department of Energy and Technology at the Swedish University of Agricultural Sciences (SLU; Uppsala, Sweden) and were performed at the Swedish Machinery Testing Institute ("Svensk Maskinprovning") (SMP; Umeå, Sweden). Data were collected during devoted days of experiments while carrying out field operations.

The tractor made available from SMP for the tests was a Valtra N101 (maximum power Pm_{MAX} = 82 kW) (**Figure 9**), equipped with the following instrumentation:

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[5] Lovarelli, D., Fiala, M., Larsson, G. (2018). Fuel and engine emissions during on-field tractor activity: a possible improving strategy for the environmental load of agricultural mechanisation. *Under review.*

- GPS (Global Positioning System), to recognise the position on field and used for the guidance control, of which the tractor was also equipped;
- CAN-bus (Controller Area Network), present on the tractor for the communication among the engine and devices of the tractor;
- personal computer with installed the Dewesoft® software, a data logger necessary to receive and store CAN-bus data and gather information on-board thanks to the user-friendly interface;
- exhaust gases analyser, a portable instrument by Testo® to measure exhaust gases emissions.

Figure 9. Tractor Valtra N101. [Source: picture by D. Lovarelli]

5.2.1 Electronics in agriculture

GPS

An already globally widespread technology is the GPS (Global Positioning System), which is used in several sectors with a multifunctional use related to the identification of geographical positions. In the agricultural sector, it has become essential in precision agriculture to map the movements of machinery and to attribute a geographical position on field to any measured data (e.g., crops presence, fertilisers distribution maps, irrigation and water stress maps, yield maps).

GPS used during this study was characterised by less than 10 cm error. CAN-bus and Testo® detected engine, tractor and emissions data, which were linked, thanks to GPS, to a geographical coordinate and, consequently, to a position on field. This position was recognised and attributed to a working state. With the processing attributing positions to the parameters permitted to evaluate their role on field, improve the efficiency of inputs use and the environmental sustainability assessment of the field operation.

CAN-bus

CAN-bus (Controller Area Network) is a serial high-speed wired data network connection commonly present on modern tractors that permits to electronic devices to communicate with each other and that, coupled with storing instrumentation, permits to collect data directly deriving from the tractor working on field and with a very detailed time scale (Speckmann and Jahns, 1999). Robert Bosch GmbH introduced it in 1986, firstly for an automotive application. Applied to agricultural tractors, is normed with SAE J1939 that defines the connections of electronic devices installed on machinery and also with ISO 11898 (ISO, 2003) standard protocol.

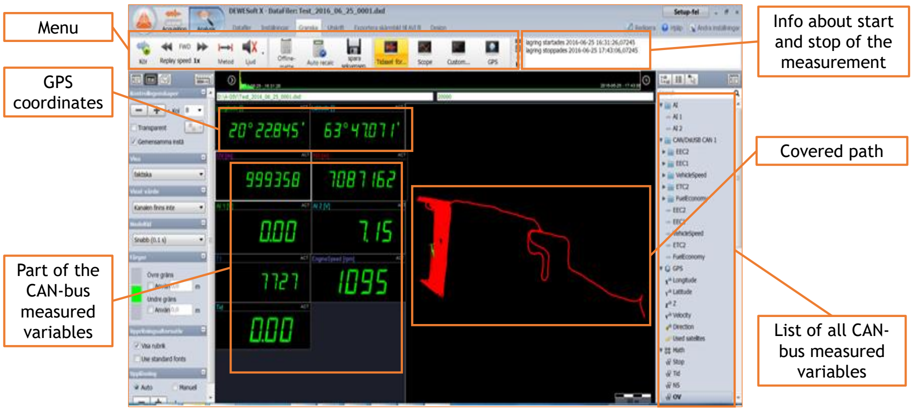
CAN-bus is a desirable system to understand and study the activity of the tractor engine and of the related devices employable while working on field. It has permitted to use and take advantage of electronics on agricultural machinery by:

- continuously monitoring the operation on field and collecting the data;
- controlling and potentially permitting to reduce inputs introduction and to ameliorate inputs efficiency and the environmental load of field operations;
- possibly developing prevision models with high precision thanks to the detailed temporal scale of collected data with which to validate models.

In addition to environmental sustainability goals, electronics can be used for other several reasons, such as diagnostics on board, maintenance scheduling, precision agriculture, etc.

Dewesoft® software

Dewesoft® is a free software used to acquire data directly from CAN-bus and store them. It is helpful both for checking on-board how variables are changing (**Figure 10**) as well as for collecting data and saving them for a subsequent processing. Dewesoft® was born to support tests and measurement contexts working with multiple applications, among which the automotive sector.



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lagring stoppades 2016-06-25 17:43:06,07245

Figure 10. On-board interface of Dewesoft® software for the collection and storage of CAN-bus data. [Source: picture by D. Lovarelli]

During the measurements, it was installed on a personal computer positioned on-board. The computer was connected through cables with the CAN-bus. By starting the data acquisition with Dewesoft®, the CAN-bus data were registered; the software showed on the display information related to:

- the start and stop of the measurement and its total duration;
- the path covered by the tractor from the start to the stop of measures;
- the coordinates of the location (latitude and longitude) expressed as UTM coordinates;
- the variables of which keeping direct monitoring during the operation.

The variables directly monitored from the main area of the screen could be selected by the user from a list of all the measured variables (more than a hundred). Moreover, the software included the possibility of implementing functions, in case the user needed information quantifiable mathematically from the measured variables.

Exhaust gases emissions analyser

The instrument for measuring engine exhaust gases is Testo® 350. It analyses the flux of gases from the exhaust pipe of the Valtra N101 tractor and results the values in ppm (in % of O_2 volume only for CO_2). It is a portable instrument, therefore not compulsorily present on board.

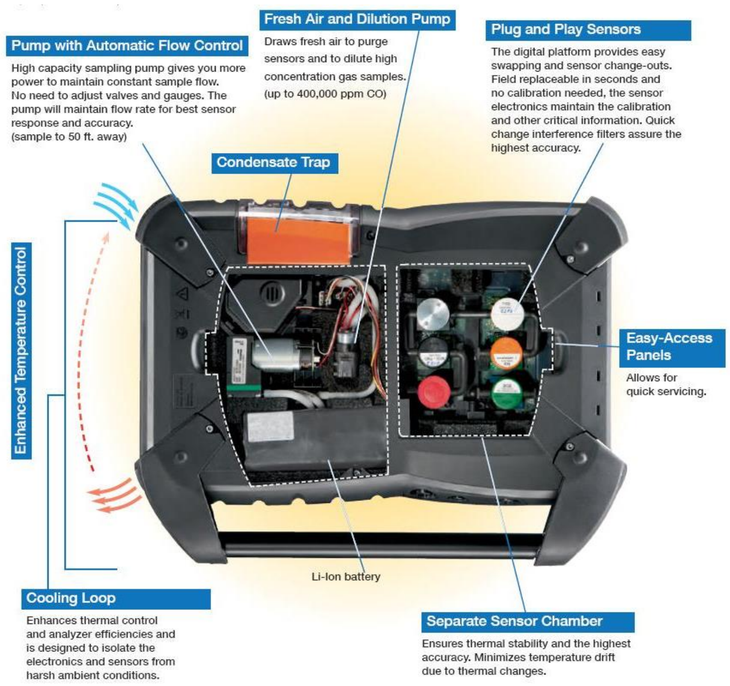
On the tractor, a system to collect exhaust gases must be present, as shown in **Figure 11**.

Figure 11. Part of Valtra N101 with the implemented sampling probe to collect exhaust gases and Testo® 350 on board. [Source: pictures by H. Harvidsson and D. Lovarelli]

This system is a stainless-steel gas sampling probe equipped with integrated thermocouples and located close to the exhaust pipe. The sampling probe works till a maximum temperature of 932°C and has a hose diameter of 8.1 mm and a standard hose length of 2130 mm. The gases reach the instrument on board, which analyses the flux (**Figure 11**) and stores data. The instrument is equipped with electrochemical and infrared sensors (up to 6 sensors) that permit the analyses, and with a chiller that removes moisture. Every 30 minutes the analyser rinses the sensors and the analysis chamber from moisture for a period of approximately 7 minutes. During this time, no emission measurement takes place.

The analysed exhaust gases with a 1 Hz frequency are: NO_X, NO, NO_2, CO and O_2. CO_2 was derived stoichiometrically from O_2 concentration. In addition, in order to keep control of the variables affecting the measure, the sample exhaust gas temperature (°C), the sample flow of exhaust gas (which is maintained as constant as possible by a pump to get the best response and accuracy from the sensors) and the Testo® temperature (°C) are also measured.

In **Figure 12** is shown Testo® instrument with a brief explanation of the main components and components' scope. With it is also available a software with which the data stored on the instrument are transferred through a cable to Testo® software for the data processing directly on the software (i.e. by tables and graphs) and for data saving in Excel format for further analyses.



Pump with Automatic Flow Control

High capacity sampling pump gives you more power to maintain constant sample flow. No need to adjust valves and gauges. The pump will maintain flow rate for best sensor response and accuracy. (sample to 50 ft. away)

Fresh Air and Dilution Pump

Draws fresh air to purge sensors and to dilute high concentration gas samples. (up to 400,000 ppm CO)

Plug and Play Sensors

The digital platform provides easy swapping and sensor change-outs. Field replaceable in seconds and no calibration needed, the sensor electronics maintain the calibration and other critical information. Quick change interference filters assure the highest accuracy.

Condensate Trap

Easy-Access Panels

Allows for quick servicing.

Cooling Loop

Enhances thermal control and analyzer efficiencies and is designed to isolate the electronics and sensors from harsh ambient conditions.

Separate Sensor Chamber

Ensures thermal stability and the highest accuracy. Minimizes temperature drift due to thermal changes.

Figure 12. Testo® instrument and scope of the main components. [Source: Testo® 350 brochure]

5.2.2 Goal and description of the field tests

The main goal of the measurements was to collect information about the variables that affect the tractor engine and the working variables, directly while carrying out the operations on field and to use them to improve the LCI (Life Cycle Inventory).

The second goal was to distinguish the data according to the working state to which they belonged in order to understand if and how the working states affect the engine behaviour, fuel consumption and exhaust gases emissions.

In other words, similarly to what was meant in ENVIAM (Lovarelli et al., 2016a, 2017), each operation was built as sum of single working states. The identified working states are:

- effective work on field;
- turns at the headlands;
- stops, due to maintenance on field, coupling/uncoupling of implements, refilling/emptying, machinery setting with the working layout, control, cleaning and maintenance on field;
- transport from farm to field and vice versa.

Since the number of variables that affect the fieldwork is huge, an experimental plan was built. It involved firstly the definition of the parameters to study and secondly the definition step-by-step of the fieldwork organisation. The identified parameters to vary along the operations were:

- working speed (km/h);
- engine speed (rpm);
- PTO speed (rpm; whether foreseen);
- working depth (cm; whether foreseen);
- headland strategy with which performing the turns.

The field tests were carried out in October 2016 on two fields located in the surroundings of the Swedish Machinery Testing Institute (Umeå, Sweden), which is the experimental testing company for machinery where the machinery and instruments were made available.

The first field was located at 2 km from the company, while the second at 9 km distance. The field texture was sandy-loamy on both fields.

On the first field, rotary harrowing was performed the first day of measurements; on the second, all the remaining operations were carried out during the following experimental days.

As shown in the engine curve in **Figure 13**, Valtra N101 tractor 4WD has maximum power Pm_{MAX} = 82 kW at the engine speed s = 1860 rpm, max torque M_{MAX} = 491 Nm and max engine speed s_{MAX} = 2400 rpm.

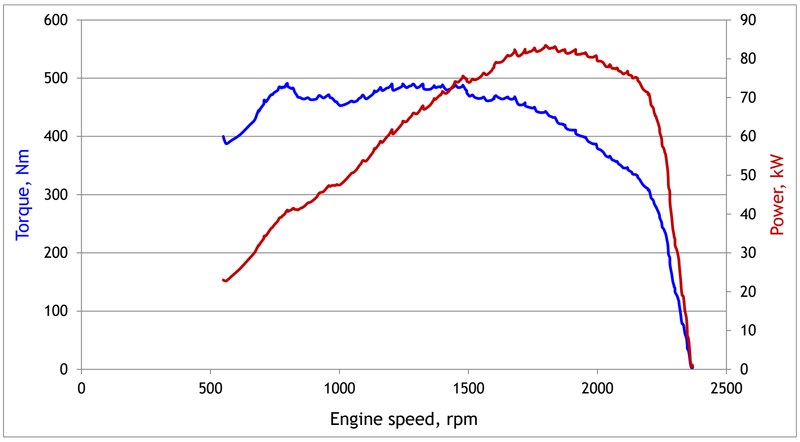


Figure 13. Engine curve of tractor Valtra N101.

The tractor belongs to the Emissive Stage IIIA (EU Directive 97/68/EC; and following amending ones: Directive 2010/26/EU, Directive 2010/22/EU) and is equipped with an Exhaust Gas Recirculation (EGR) system that reduces NO_X formation in accordance with the EU Directive.

The EGR is a technique used for steeply reducing NO_X formation in internal combustion engines. It involves (i) recirculating a portion of engine exhaust gases from the exhausts pipe to the engine cylinders, (ii) permitting a dilution of O_2 in the incoming air stream, and (iii) providing gases inert to combustion to act as absorbents of combustion heat in order to reduce the peak in-cylinder temperatures. Since NOx are produced in a narrow band of high cylinder temperatures and pressures, effecting these conditions brings to lowering NO_X exhaust emission formation up to 90%.

Regarding the operations studied, the following implements were accessible:

(i) ploughing: 3-furrows mouldboard plough;
(ii) harrowing: rotary harrow, 3.0 m wide;
(iii) harrowing: spike harrow, 6.0 m wide;
(iv) sowing: universal sowing machine, 6.0 m wide;
(v) rolling: compact roller, 5.4 m wide.

Day 1: rotary harrow

The first operation performed was the rotary harrowing, with which were compared different headland strategies.

The working parameters (e.g., working speed, working depth) were kept constant during almost all phases of effective work (as explained in detail in *Paper 5*). The field shape was 100 m wide and 170 m long and the field was split as shown in **Figure 14**.

The worked area was 1.7 ha; the total working time on field was 2.95 h.

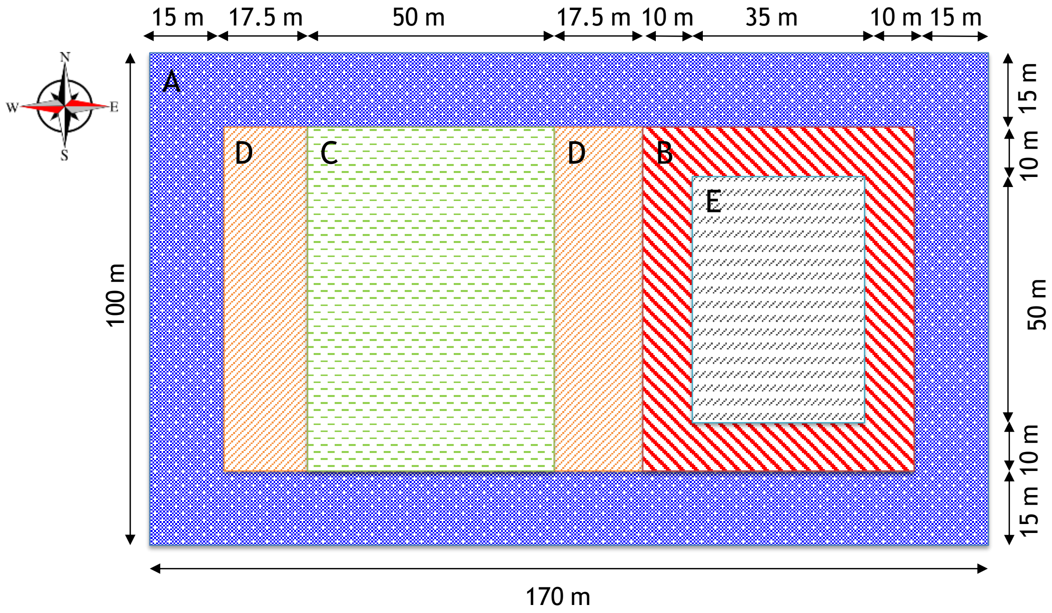


Figure 14. Field organisation for the rotary harrow operation.

Letters A-E present in the figure correspond to the different headland strategy identified. The distinction among them is illustrated in **Figure 15**.

As mentioned above, although the effective working conditions were kept constant along the whole operation, during strategy D, the working speed during the upstream effective work (North-South) was varied respect to the one downstream (South-North) (fast gear 3 and fast gear 1, respectively). Additionally, in the strategy C, half field was performed with engine speed = 1700 rpm and the second part with engine speed = 2000 rpm. This was aimed to collect useful data related to the effective work on the rows.

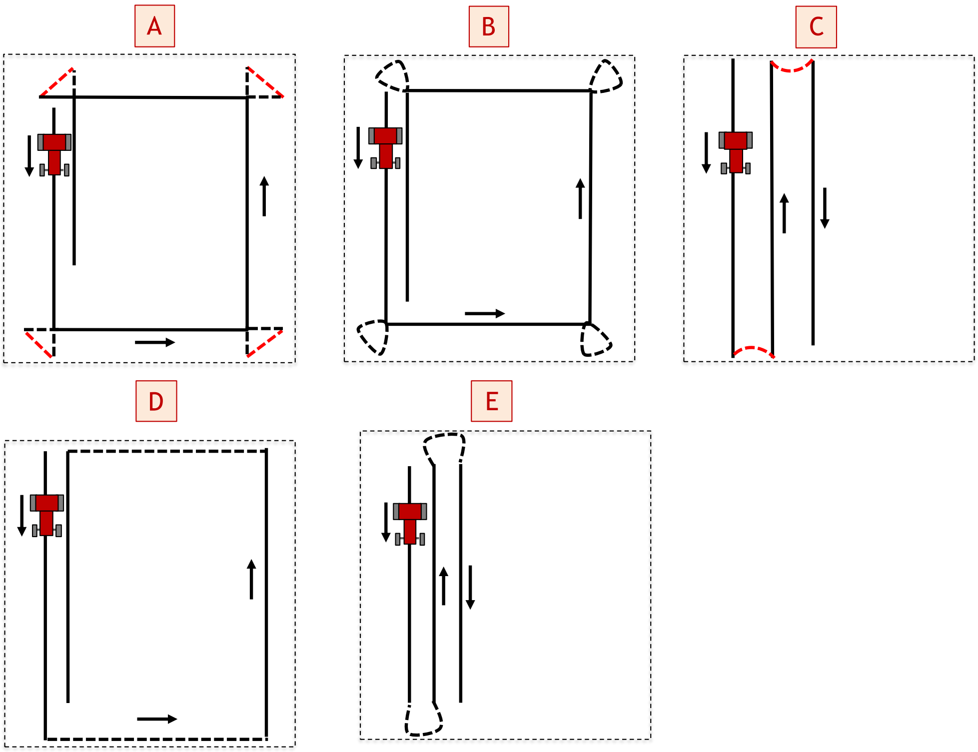


Figure 15. Alternative headland strategies. The spotted lines identify the turn on the headlands with the black-coloured line for the forward direction and the red-coloured line for the reversing.

The decision about where to keep the working variables constant or not was dependent on practical reasons concerning the field shape and, mainly, the disposition on field of the studied alternatives. In more details:

- option A was completed on the field boundary;
- options B, C and D were performed overall the available field length;
- option E was performed in the inner part of the area remaining available from the one worked with option B.

Day 2: plough

The field with the 3-furrow mouldboard plough was split as shown in **Figure 16**. The differences among the sections were due to:

(i) two engine speed ranges: s_A = 1400 rpm; s_B = 1800 rpm;
(ii) two working speeds: v_A = 5 km/h; v_B = 7 km/h;
(iii) two working depths: H_A = 18 cm; H_B = 28 cm.

The worked area was 1.2 ha and the total working time on field was 2.94 h.

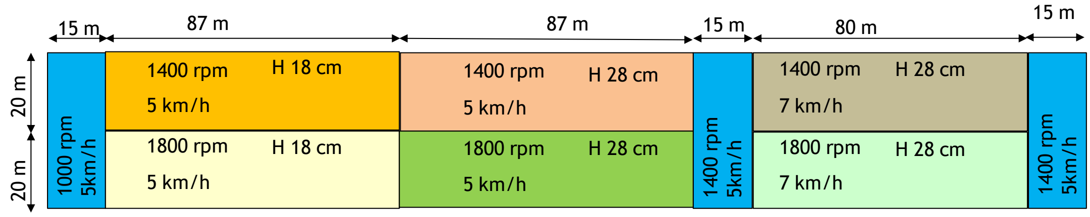


Figure 16. Field distinction during ploughing. Blue sections identify the headlands (3 headlands).

Day 3: spike harrow

Harrowing was performed splitting firstly the field in two sections: the first section (A; top) with a working depth H_A = 12 cm and the second (B; middle and bottom) with a working depth H_B = 8 cm. Section A was spilt in three parts on the length, where different working speed and engine speed were studied as shown in **Figure 17**. Section B, instead, was split in three parts on the width in order to avoid disturbances on the length side for the exhaust emissions measurement. The three sub-sections were characterised by three different engine speeds and two different working speeds.

The worked area was 4.2 ha and the working time on field was 3.41 h.

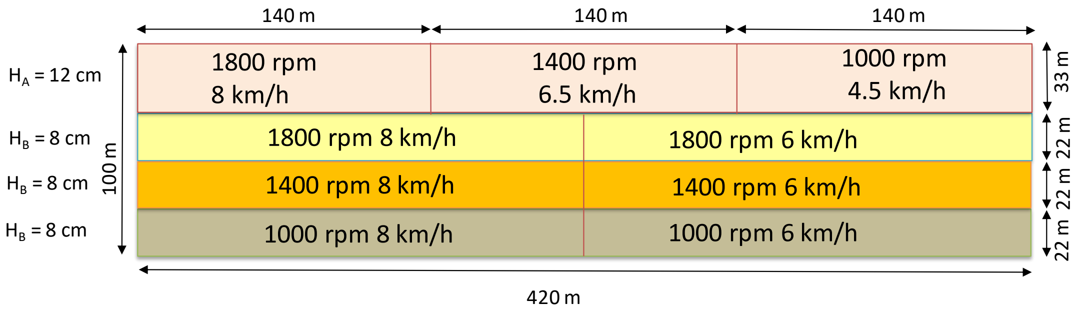


Figure 17. Field distinction during harrowing.

Day 3: sower

The field was split in sections according to two different headland strategies, one on the outer part (strategy "A" according to previous Figure 15) and another on the inner part (strategy "E" according to Figure 15).

Operatively, as reported in **Figure 18** the differences between the two main areas considered are: (i) working speed: v_1 = 5 km/h, v_2 = 8 km/h, (ii) PTO speed: PTO_1 = 1000 rpm, PTO_2 = 540 rpm, (iii) engine speed s_A = 1080 rpm and s_B = 1800 rpm.

The sowing was performed with a universal mechanical seeder on an area of 4.2 ha; the working time on field was 1.2 h.

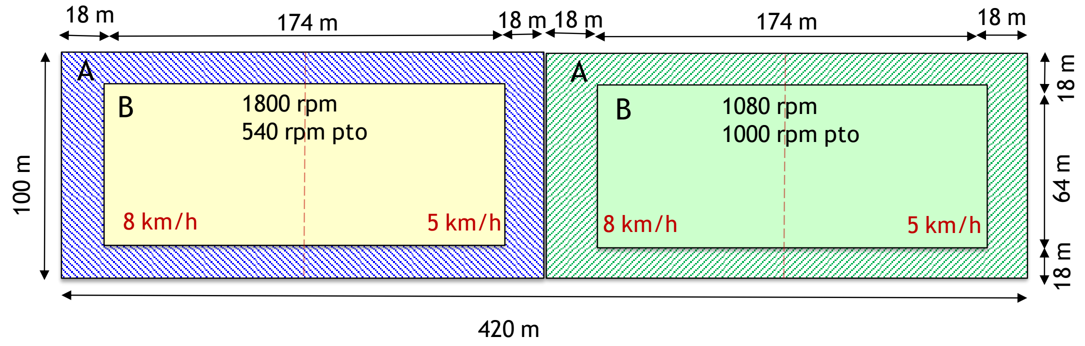


Figure 18. Field distinction during sowing.

Day 4: roller

For rolling (**Figure 19**), only the working speed was changed: v_A = 7 km/h and v_B = 10 km/h. The operation with the compact roller was performed on the same field of former operations, avoiding one third of the field (worked area: A = 2.78 ha). The total working time on field was 0.41 h.

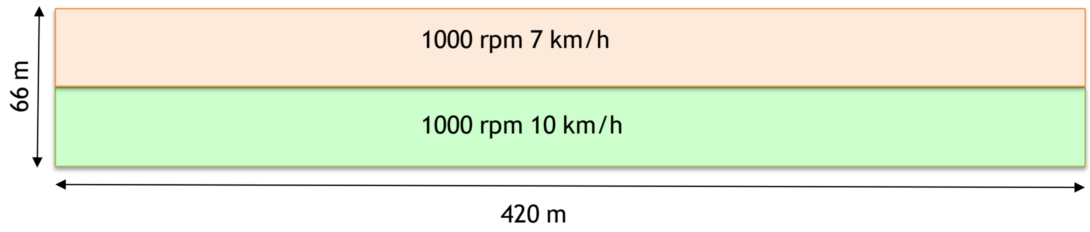


Figure 19. Field distinction during rolling.

5.2.3 Data processing

Once collected the data, they were exported as MS Office Excel 2013 files.

First, GPS coordinates were used to build the path followed by the tractor from the SMP centre to the field and vice versa and during the work on field. Moreover, as shown in **Figure 20** as an example, the geographical coordinates were used to distinguish automatically the working states:

- when the GPS coordinate varied without exceeding a defined angle, the tractor was working on the row/stretch (effective work),
- when the GPS coordinate varied according to a defined angle, the tractor was turning,
- when the GPS coordinate did not change, the tractor was stopping.

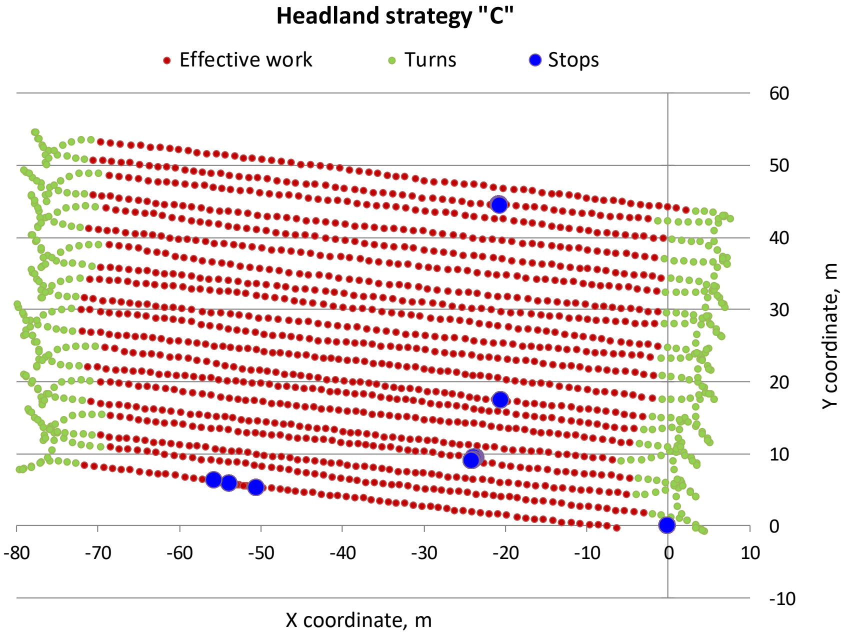


Figure 20. Description of the field with the GPS coordinates distinguishing among rows of effective work (red), turns at headlands (green) and stops (blue).

Another preliminary step included the offsetting of CAN-bus/Dewesoft® data with Testo® ones. In fact, according to the switch on of the instruments, a delay between the starting point of measures was unavoidable. Therefore, they were offset by plotting fuel consumption (CAN-bus/Dewesoft®) with CO_2 (Testo®), since these two variables are strongly correlated with each other ($R^2 = 0.90$-0.93, according to the cases).

After this step, every measured variable was grouped and linked with its spatial position and working state, which allowed both for realising graphs of fields with variables grouped in ranges (**Figure 21**) and relating the variables to the working state (**Figure 22**). The engine load ranges are: (i) 0%-20%; (ii) 20%-40%; (iii) 40%-60%; (iv) 60%-80%; (v) 80%-100%.

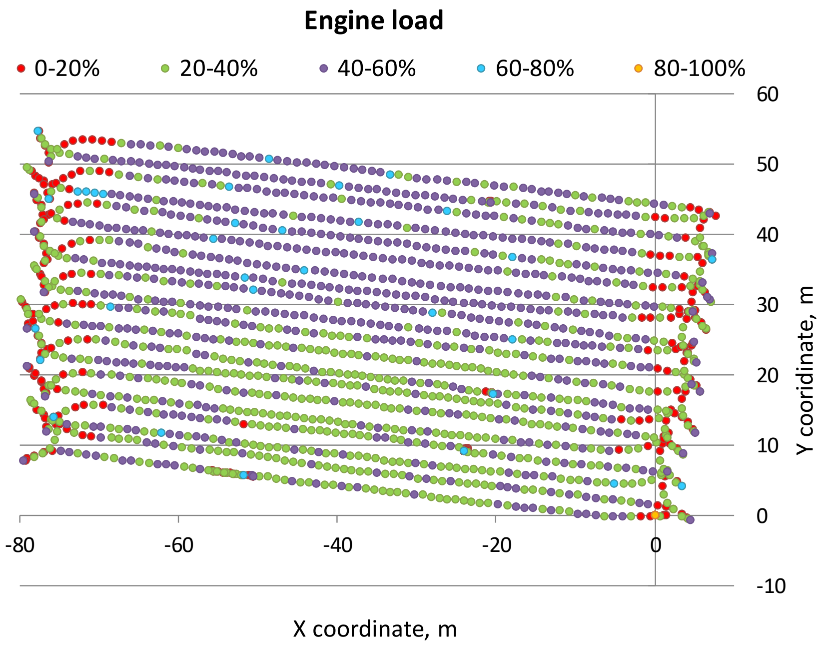


Figure 21. Example of field defined by the ranges in engine load (%).

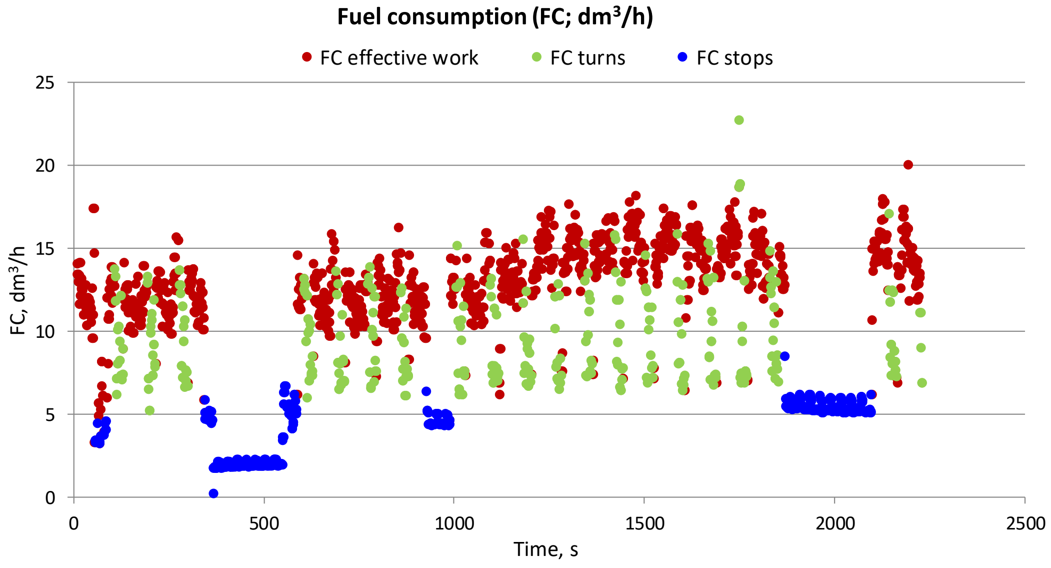


Figure 22. Fuel consumption for one of the operations distinguishing the spatial groupings (effective work on rows, turns at headland, stationary/stops).

To characterise every row and turn, each of them was numbered and an average value for all variables that described every row, turn and stop was calculated.

Values for fuel (kg/h) and emissions (g/h) were also quantified in specific values (g/kWh) by dividing the hourly consumption/emission (g/h) by the developed engine power (kW), as shown in the example of **Figure 23**. Thus, comparisons could be performed among working states and literature data.

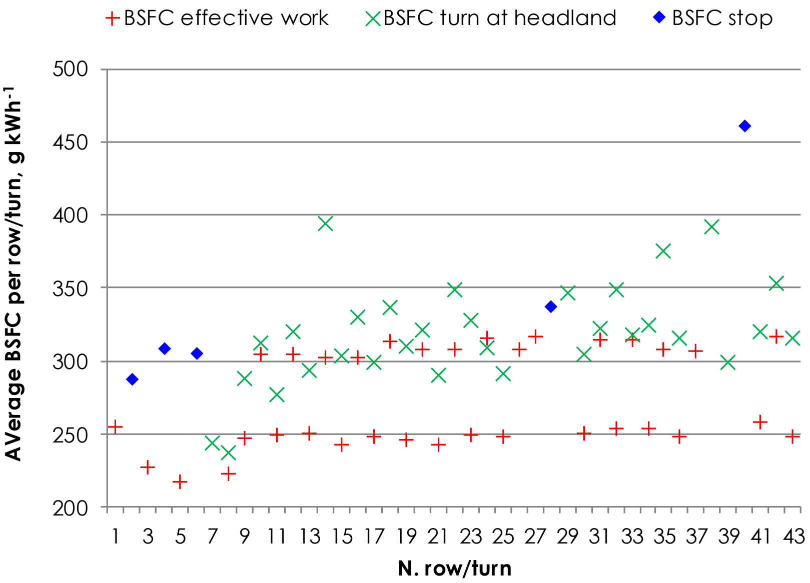


Figure 23. Average brake specific fuel consumption (bsfc; g/kWh) per row of effective work (red), per turn (green) and stop (blue) for the analysed operation.

Regarding the prediction of fuel consumption and exhaust gases emissions, the model developed by Speckmann and Jahns (1999) and further used by Lindgren (2005) was adopted.

This model describes the steady-state condition and considers several variables, of which engine speed (s; rpm) and torque (M; Nm) represent the most important data to have from measurements. Nine engine-specific coefficients are also needed for the modelling and they were identified for the tractor engine using the least square fit with Matlab® software (*Paper 5*).

In the case study by Lindgren (2005) is stated that the average difference between recorded and calculated value with the model for the steady-state conditions was approximately 30%. In this study, however, it was less than 20%, mainly due to the fact that the operations' working characteristics were previously defined and respected reducing the presence of transients. A second reason is that in Lindgren (2005), several operations with not negligible transients were assessed (e.g., loading operation). On the opposite, in this case study, operations were characterised by transients, but with not as much relevance.

5.2.4 Model adopted for fuel consumption and engine exhaust emissions

The model by Lindgren (2004) is described by Equation (7):

$$FC = c_1 \cdot s + c_2 \cdot s^2 + c_3 \cdot s^3 + M \cdot (c_4 \cdot s + c_5 \cdot s^2 + c_6 \cdot s^3) + M^2 \cdot (c_7 \cdot s + c_8 \cdot s^2 + c_9 \cdot s^3) \tag{7}$$

where:

FC = fuel consumption (dm^3/h);

$c_1..c_9$ = engine-specific coefficients;

s = engine speed (revolutions/min);

M = engine torque (Nm).

The same equation, corrected with other coefficients (c_1-c_9) is also used for quantifying the emission of CO_2, CO and NO_X.

Equation (7) predicts with significance fuel consumption and engine emissions when there is no fast change in engine speed and/or in torque. When such transient effects play a prominent role, a second part of equation that considers 3 more engine-specific coefficients describing the transients can be introduced (Lindgren, 2004). However, in this PhD Thesis this second equation was studied, but not applied because of the restrained role of transients (*Paper 5*).

Differently from the models adopted in ENVIAM for fuel and emissions, in this case results are strictly related to the specific engine studied. Therefore, for what regards engine emissions, no additional consideration is needed on Emissive Stages.

5.3 WF

The WF was studied within the WFN framework and, after having evaluated the main critics about the methodological assessment, a proposal for improving the indicator was made.

5.3.1 Studies set-up

The presence of a wide literature often in disagreement or inconsistent within itself brought to the need of a review of studies on WF carried out about agricultural products and referred to the period 2000-2015, as described in *Paper 6*[6].

In more details, this period corresponds with the development of the WF indicator in accordance with the WFN guidelines and, in the second part of this period, with the evidence of important critics and the introduction of ISO 14046 (2014). For new practitioners, this period resulted quite chaotic and needed clarity.

From the literature analysis with more than 90 scientific papers on agricultural productions, some major methodological concerns emerged. The review was organised focusing on three main topics:

- early studies related to a worldwide scale with a general analysis of numerous agricultural products;
- studies related to national/local agricultural products (food and feed);
- studies related to bioenergy production from the agricultural sector.

The methodological drawbacks and lacks in the assessment that emerged from this study were further deepened in *Paper 7*[7], where were suggested some methodological changes as schematically shown in **Figure 24**:

- considering grey water, the effect of all pollutants introduced in the water system was evaluated by means of LCA, taking into consideration both the impact categories determined by water issues (eutrophication and ecotoxicity) and the WFgrey in a new index called Pollution Water Indicator (PWI);
- considering blue water, the gross volume for irrigation (Water Footprint Applied, WFA) was adopted in accordance with García-Morillo et al. (2015), taking into account the irrigation technology and its efficiency and the irrigation turn.

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[6] Lovarelli, D., Bacenetti, J., Fiala, M. (2016b). Water Footprint of crop productions: A review. Science of the Total Environment. 548-549, 236–251.

[7] Lovarelli, D., Ingrao, C., Fiala, M., Bacenetti, J. (2016c). Beyond the Water Footprint: a new framework proposal to assess freshwater environmental impact and consumption. Journal of Cleaner Production. 1-11.

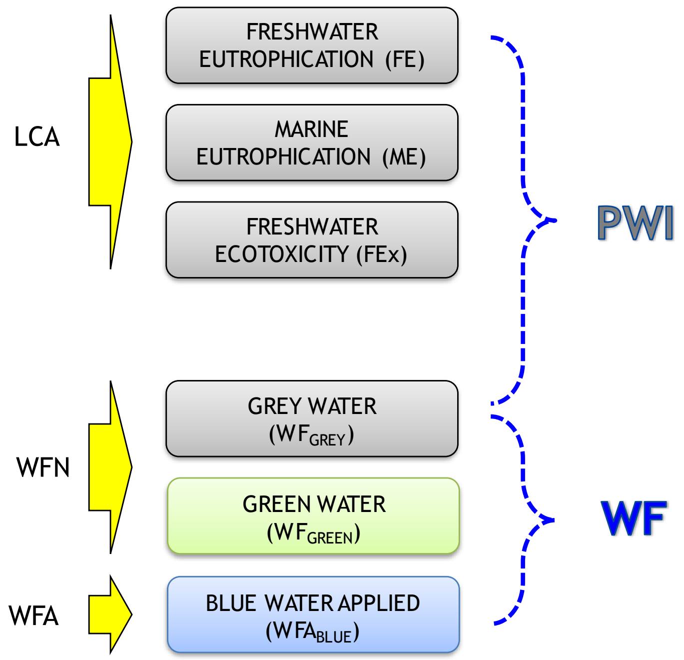


Figure 24. Top: Pollution Water Indicator (PWI) assessed considering Freshwater Eutrophication, Marine Eutrophication and Freshwater Ecotoxicity impact categories (evaluated by means of LCA method) and WFgrey (quantified with the WFN approach). Bottom: WF assessed considering the WFN for grey and green water and the Water Footprint Applied (WFA) for blue water.

5.3.2 Pollution Water Indicator (PWI)

Regarding the critic on WFgrey, in order to evaluate the environmental effect of all the pollutants effectively introduced in the system during production, the Pollution Water Indicator (PWI) was developed.

As illustrated in **Figure 25**, the PWI is a numerical value calculated as the area of a rhombus, whose vertexes are made of WFgrey and of three environmental impact categories quantified by means of LCA that play a role on the water system, which are:

- Freshwater Eutrophication (FE; kg Peq),
- Marine Eutrophication (ME; kg Neq),
- Freshwater Ecotoxicity (FEx; CTUe).

These categories belong to the composite method recommended by the International Reference Life Cycle Data System ILCD (ILCD Handbook, 2011).

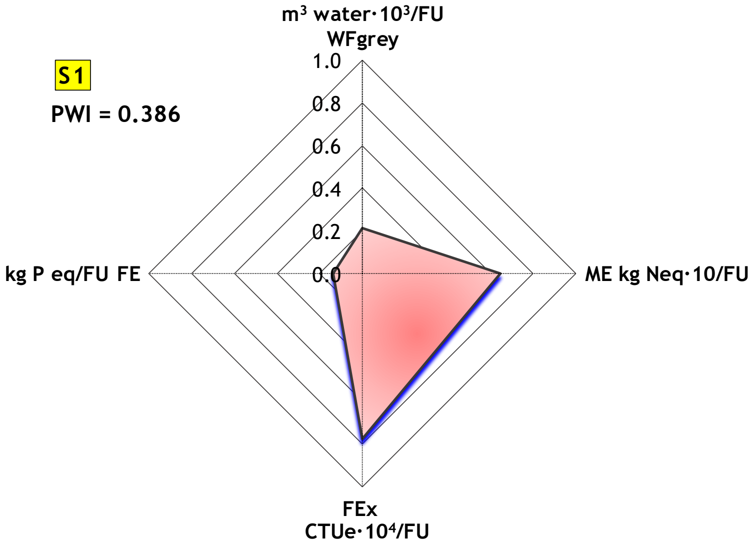


Figure 25. Pollution Water Indicator (PWI) for a studied scenario as example.

The value is obtained considering variables with a different unit of measure, but the variables are indexed and referred to the same scale (0-1) and all of them are meant to weigh the same on the outcoming result. This makes the PWI an applicable indicator.

The impact categories were selected because they describe mostly the leaching and runoff of nutrients (nitrogen and phosphorous compounds) and of chemical products (pesticides, herbicides, fungicides) sprayed during agricultural cultivation phases. In particular, with FE and ME are analysed the P eq. and N eq. compounds, respectively, and their environmental role on water; with FEx is studied the toxic effect of chemicals application and, mainly of their leaching to water.

These categories, however, were put together with WFgrey in order to identify whether WFgrey was effectively representative for the most important pollutant evaluated within its methodology and to study if a better indicator could be identified for water quality evaluations.

6. Results and discussion

6.1 LCA and mechanisation alternatives: case studies for agricultural machinery field operations

6.1.1 Soil tillage (Paper 1-2-3)

The use of ENVIAM as data source for fulfilling the inventory showed that variables such as soil texture, field shape, implement dimension and tractor engine power are fundamental to assess the environmental impact of agricultural machinery field operations.

Table 1. Working time and engine load of the 3-furrows mouldboard ploughing operation studied in *Paper 1*.

Working time*		Unit of measure	TEF	TAS	TPH	TM	TT
		%	76	19	2	1	2
Tractor A (sandy soil)	Average engine load	%	73	30	2	2	40
	Engine Power	kW	58				
	Fuel consumption	kg/ha	16.7				
	Emissions: (a) CO, (b) HC, (c) NO_X, (d) PM, (e) CO_2	g/ha	(a) 423.5, (b) 16.5, (c) 264.3, (d) 2.0, (e) 52,492				
Tractor B (medium textured soil)	Average engine load	%	80	30	2	2	40
	Engine Power	kW	96				
	Fuel consumption	kg/ha	25.7				
	Emissions: (a) CO, (b) HC, (c) NO_X, (d) PM, (e) CO_2	g/ha	(a) 914.5, (b) 35.6, (c) 570.8, (d) 4.4, (e) 80,815				
Tractor C (clayey soil)	Average engine load	%	80	30	2	2	40
	Engine Power	kW	197				
	Fuel consumption	kg/ha	48.0				
	Emissions: (a) CO, (b) HC, (c) NO_X, (d) PM, (e) CO_2	g/ha	(a) 1265.0, (b) 70.3, (c) 683.5, (d) 8.7, (e) 151,286				

Notes: * TEF: effective work, TAS: turns at headlands, TPH: Implement arrangement on farm, TM: Field maintenance, TT: transfer field-farm and vice versa.

In **Table 1** are reported the main inventory results that characterised the field operation of ploughing studied in *Paper 1*; the outcomes are assessed as sum of different timings, each of which characterised by defined engine working features (mainly, of the engine load). In particular, engine load during TEF was calculated in accordance with the calculations related to tractor-implement coupling. Therefore, according to the working assumptions (e.g., soil texture, soil coefficients, implement features) the resulting calculated power that had to be developed by the tractor was used to select a "real" tractor available on the market and present in the database, characterised by a sufficient engine power to carry out the operation. Hence, the ratio between the tractor developed engine power (calculated during TEF) and the tractor maximum engine power (rated power) define the engine load during TEF. On the sandy soil, this resulted in a value equal to 73%, whereas in the cases of medium texture and clayey soils, it was equal to 80%. Mainly, this result is affected by the absorbed engine power of the operation as well as by the fact that in the reality are available several alternatives of tractors in terms of power, but according to the pedo-geographic context and farm activity it cannot be assumed that the optimal tractor for every farmer is available on the market and/or on farm. In fact, in order to avoid wrong environmental assessments, the mechanical choices must describe the real working context. Regarding the other timings, for which commonly the tractor is idling or almost idling (TAS, TPH, TM) and the transport (TT), the average engine load is assumed taking into consideration the working conditions of the study area.

From the table, it can be seen that the inventory values (i.e. fuel and engine emissions) are widely different in the different work conditions. The tractor needed to carry out the operation is extremely different: 58 kW, 96 kW and 197 kW, respectively for the sandy, medium texture and clayey soil cases. The fuel consumption during ploughing ranges between 16.0 kg/ha on the sandy soil and 48.0 kg/ha on the clayey one, which is 3 times higher than the previous. Thus, when the processes are carried out in different contexts (e.g., different soil texture) and have effect on the inventory data, the resulting environmental load quantified by means of LCA will differ.

Following this analysis on the inventory of ploughing, the result on the environmental perspective came from the comparison completed in *Paper 2*, as shown in **Figure 26**. **Table 2** reports details about the studied environmental impact categories.

Table 2. Impact categories selected for the study from ILCD characterisation method.

Impact category	Symbol	Unit of measure
Climate Change	CC	kg CO_2 eq
Ozone Depletion	OD	kg CFC-11 eq
Particulate Matter Formation	PM	kg PM 2.5 eq
Photochemical Oxidant Formation	POF	kg NMVOC eq
Terrestrial Acidification	AC	molc H+ eq
Terrestrial Eutrophication	TE	molc N eq
Freshwater Eutrophication	FE	kg P eq
Marine Eutrophication	ME	kg N eq
Mineral, fossil and renewable resources depletion	MFRD	kg SB eq

In this study, the compared options were:

(i) ECO: Ecoinvent unmodified process, available in Simapro®,

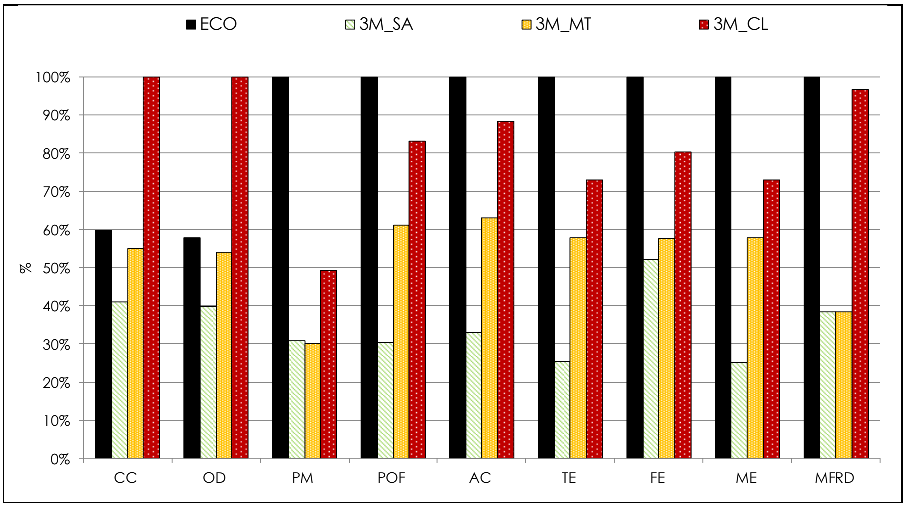
(ii) 3M_SA, 3M_MT, 3M_CL: 3-furrows mouldboard (3M) ploughing defined with ENVIAM on sandy (SA), medium texture (MT) and clayey soils (CL),

(iii) 3S_MT, 3S_CL: 3-furrows slatted (3S) ploughing defined with ENVIAM on medium texture and clayey soils (not applicable on sandy soils),

(iv) 3M_SA+, 3M_MT+, 3M_CL+: the same cases occurred at point (ii) but considering a field shape that entailed a higher working time,

(v) 5M_SA, 5M_MT, 5M_CL: the same cases occurred at point (ii) but with a 5-furrows mouldboard plough (5M) instead of a 3-furrows one.

In all the cases (except for ECO), the coupling between implement and tractor was performed with ENVIAM which permitted consistent coupling choices in terms of selection of the tractor responding to the implement and pedo-climatic-operating variables. The case ECO, instead, is the unmodified process available in LCA software (i.e. Simapro®) which is adopted commonly by users that choose an "average ploughing process".



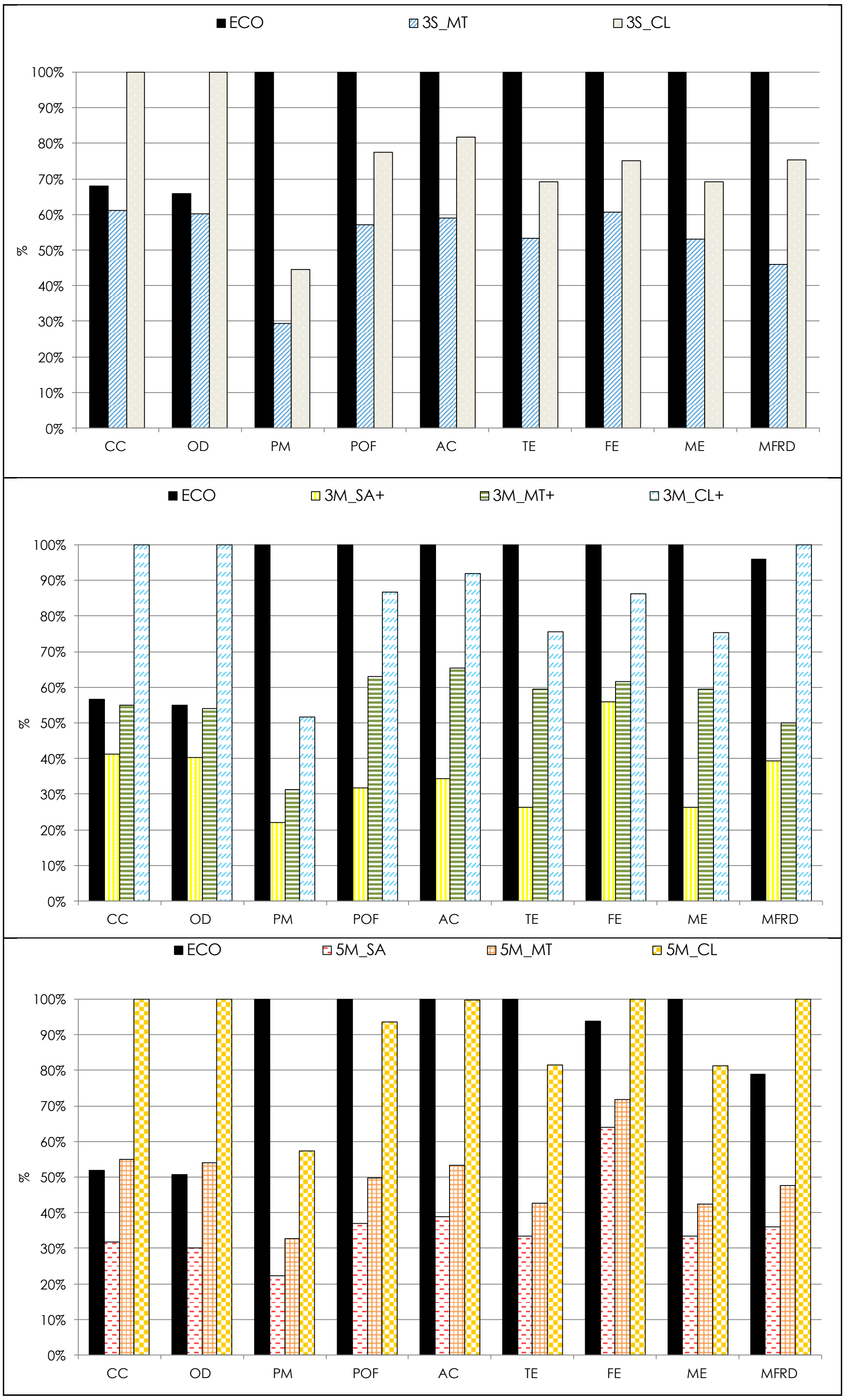


Figure 26. Environmental impact results of the compared ploughing alternatives.

The environmental impacts reported in the figure are described by putting the most impacting option at 100% of impact and all other options are expressed as a percentage respect to the worst solution. It can be observed that impacts are widely variable depending mainly on the soil texture.

Thus, adopting the average value for ploughing available in Ecoinvent database (case ECO) can be misleading in several cases, causing not negligible underestimations (e.g., if the soil texture is clayey) or overestimations (e.g., if the soil texture is sandy) of environmental impacts.

The most sustainable options are obtained on sandy soils, this not meaning at all that on sandy soils the environmental impact of a full cultivation practice is lower, but that in the studied cases the environmental impact of ploughing is lower, being even less than 60%-80% respect to ECO for most environmental impact categories. As expected, the opposite occurs on clayey soils, where higher engine power is required for the operation and where ploughing is responsible for high fuel consumption and related engine exhaust gases emissions.

On the environmental perspective, the most interesting result is how, why and on which impact categories the outcomes from Ecoinvent differ from those by ENVIAM. In fact, on PM, POF, AC, TE and ME the highest environmental load is attributed to Ecoinvent ploughing process, mainly due to the exhaust gases emissions calculation method and Emissive Stage to which tractors belonged. Additionally, these impact categories are also affected by the materials production and consumption of machinery, therefore the mass of tractor and implement consumed during the operation affects also these outcomes. In particular, the mass of the tractor is linked to its power, therefore, an "average" tractor (i.e ECO case) instead of a "real" tractor (i.e. ENVIAM cases) can make a considerable difference in terms of kg of iron, plastics, etc. On CC and OD, for which fuel is the main hotspot, the ploughing on clayey soil (3M_CL; 3S_CL; 3M_CL+; 5M_CL) showed the worst environmental outcomes, given the higher fuel production and consumption respect to the other cases. For the same reason, on FE and MFRD the most impacting solution is 5M_CL. This shows once more the considerable underestimation or overestimation of the environmental impacts attributed to the average ECO ploughing process, characterised by a fuel consumption equal to 21.7 kg/ha of fuel.

For the detailed data analysis and discussion on the full comparison among these options, see attached *Paper 2*.

When evaluating the whole seedbed preparation, the complexity in the selection of the alternatives and in the combination of implements and soil textures increases. In **Table 3a-c** is reported the list of the solutions displayed for seedbed preparation as studied in *Paper 3*. The code reported on the left of the tables uniquely describes the list of operations of primary and secondary tillage for the seedbed preparation. The most important assumption is that the operations were performed by farmers and only with their available machinery, therefore no intervention by contractors was considered. Primary tillage could be carried out also with a previous subsoiling, while secondary tillage with one or two implements and with one or more repetitions. The number of repetitions per operation can differ according to the refinement intensity.

Table 3a. Solutions studied for the seedbed preparation on a sandy soil with ploughing depth (i-a) 25 cm and with secondary tillage refinement (i-b) low (L) or (ii-b) high (H). (*) Option shown in Figure 27a.

Code	Operations	Number of repetitions		
		L25	H25	H35
A*	Mouldboard plough Spring tine harrow	1 1	1 1	1 1
B*	Mouldboard plough Fixed teeth harrow	1 1	1 1	1 1
C*	Disc plough Spring tine harrow	1 1	1 1	1 1
D*	Subsoiler Spring tine harrow	1 1	1 1	1 1

Table 3b. Solutions studied for the seedbed preparation on a medium texture soil with ploughing depth (i-a) 25 cm or (ii-a) 35 cm and with secondary tillage refinement (i-b) low (L) or (ii-b) high (H). (-) not possible option. (*) Option shown in Figure 27b.

Code	Operations	Number of repetitions		
		L25	H25	H35
A*	Mouldboard plough Spring tine harrow	1 1	1 1	1 1
C	Disc plough Spring tine harrow	1 1	1 2	1 2
D	Subsoiler Spring tine harrow	1 2	1 3	1 3
E*	Mouldboard plough Rotary harrow	1 1	1 2	1 2
F	Mouldboard plough Disc harrow	1 1	1 2	1 2
G	Subsoiler Rotary tiller	-	1 1	1 1
H*	Subsoiler Disc harrow	1 2	1 3	1 3
I	Disc plough Rotary harrow	-	1 1	1 1
J	Mouldboard plough Spring tine harrow Rotary harrow	-	1 1 1	1 1 1
K*	Slatted plough Disc harrow Spring tine harrow	-	1 1 1	1 1 1

Table 3c. Solutions studied for the seedbed preparation on a clayey soil with ploughing depth (i-a) 25 cm or (ii-a) 35 cm and with secondary tillage refinement (i-b) low (L) or (ii-b) high (H). (-) not possible option. (*) Option shown in Figure 27c.

Code	Operations	Number of repetitions		
		L25	H25	H35
A*	Mouldboard plough Spring tine harrow	1 2	1 3	1 3
E*	Mouldboard plough Rotary harrow	1 1	1 2	1 2
G	Subsoiler Rotary tiller	1 1	-	-
H	Subsoiler Disc harrow	1 2	1 3	1 3
J*	Mouldboard plough Spring tine harrow Rotary harrow	1 1 1	1 2 1	1 2 1
K*	Slatted plough Disc harrow Spring tine harrow	1 1 1	1 1 2	1 1 2
L	Subsoiler Rotary tiller Spring tine harrow	-	1 1 1	1 1 1
M	Mouldboard plough Disc harrow Rotary harrow	-	1 2 1	1 2 1

In **Table 4** are listed the studied environmental impact categories and in **Figure 27a-c** are shown the resulting environmental loads of these options.

Table 4. Impact categories selected for the study from ReCiPe characterisation method.

Impact category	**Symbol**	**Unit of measure**
Climate Change	CC	kg CO_2 eq
Ozone Depletion	OD	kg CFC-11 eq
Terrestrial Acidification	TA	kg SO_2 eq
Freshwater Eutrophication	FE	kg P eq
Marine Eutrophication	ME	kg N eq
Human Toxicity	HT	kg 1,4-DB eq
Photochemical Oxidant Formation	POF	kg NMVOC eq
Particulate Matter Formation	PM	kg PM 10 eq
Terrestrial Ecotoxicity	TEx	kg 1,4-DB eq
Freshwater Ecotoxicity	FEx	kg 1,4-DB eq
Marine Ecotoxicity	MEx	kg 1,4-DB eq
Metal depletion	MD	kg Fe eq
Fossil depletion	FD	kg oil eq

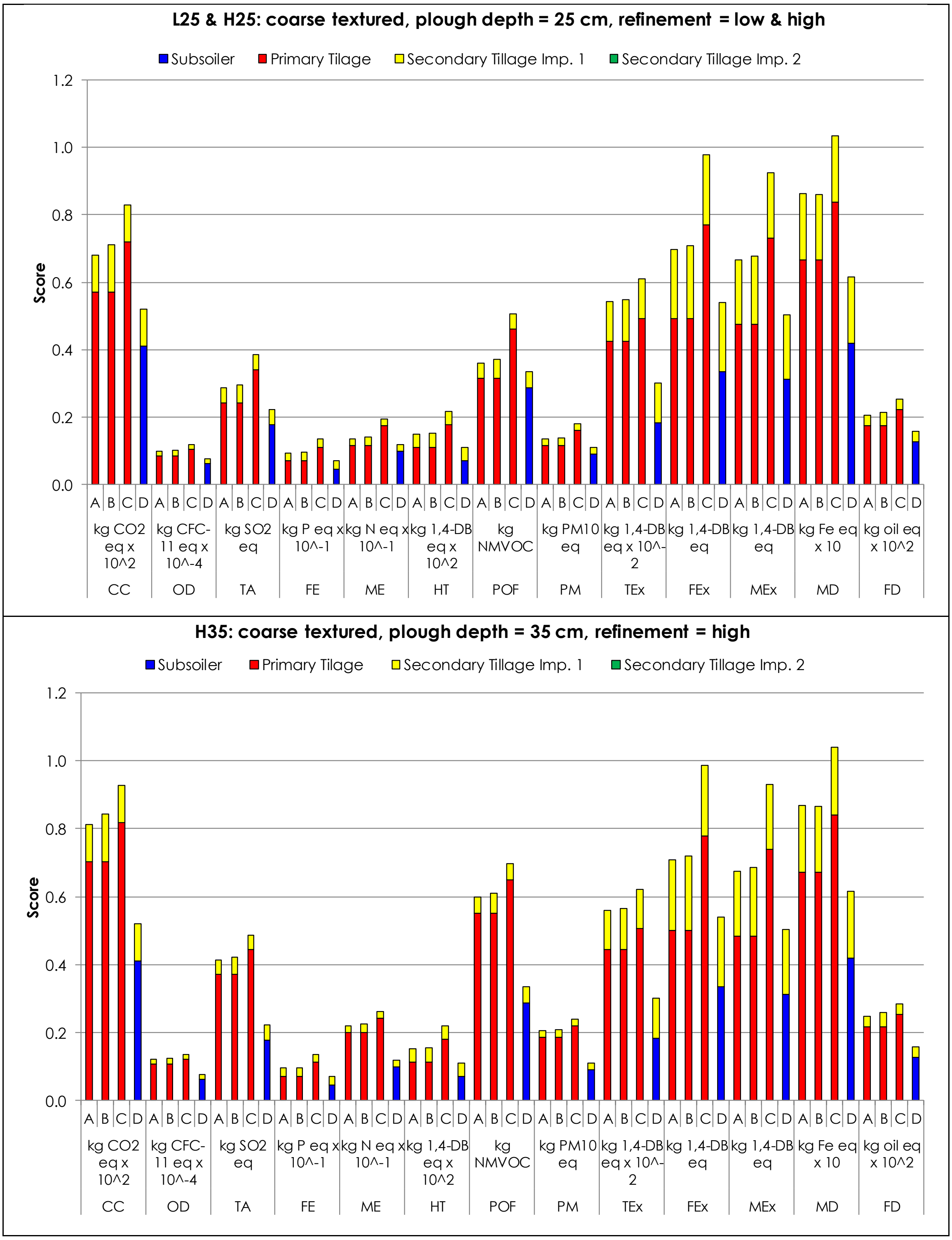
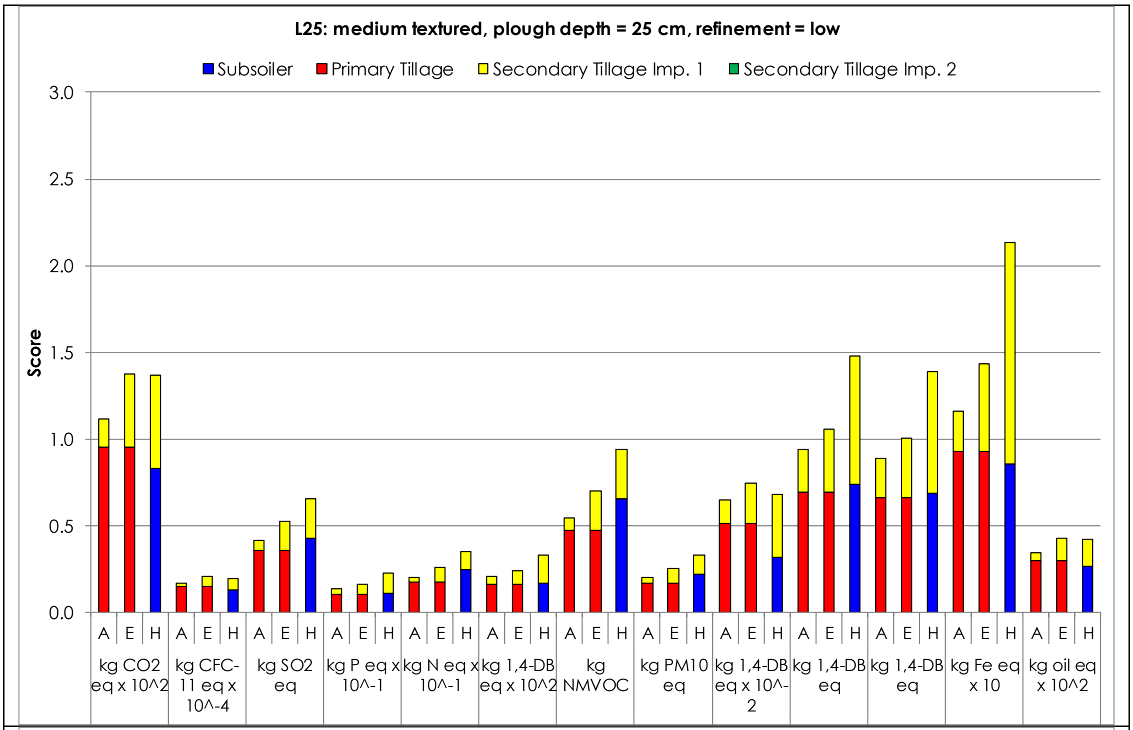
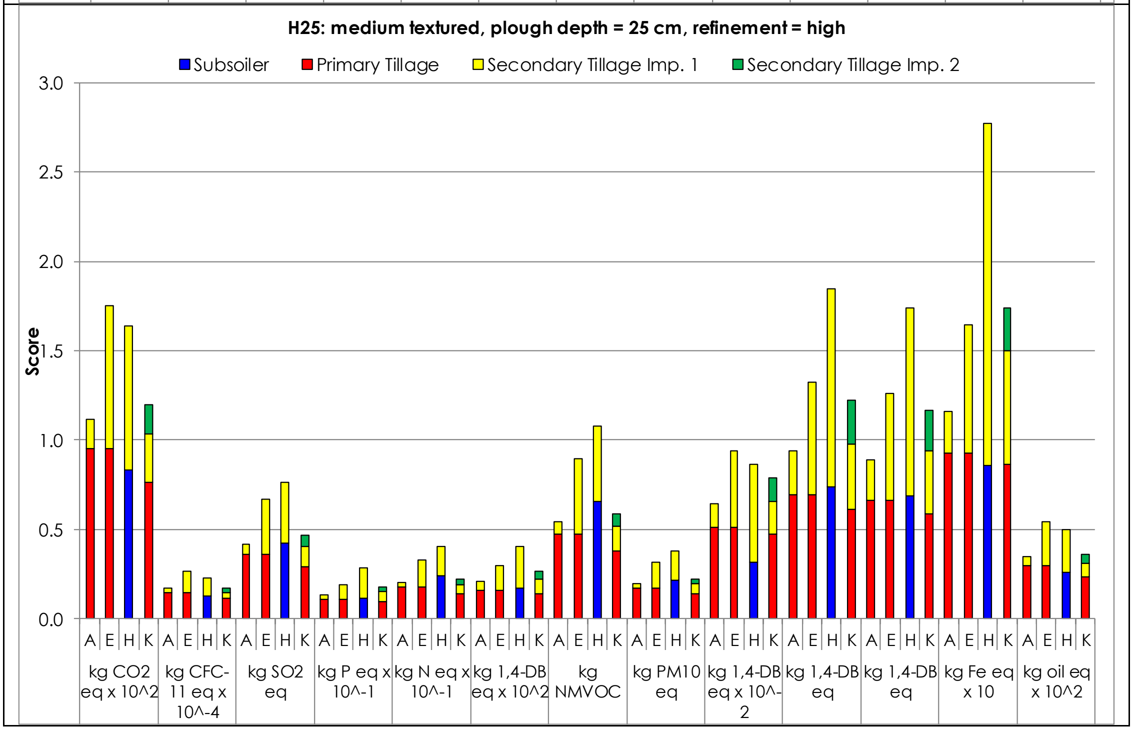


Figure 27a. Environmental results of the seedbed preparation on a sandy soil (coarse textured) with both ploughing depths (d_1 = 25 cm and d_2 = 35 cm) and low (L) or high (H) refinement during the secondary tillage.


L25: medium textured, plough depth = 25 cm, refinement = low


H25: medium textured, plough depth = 25 cm, refinement = high

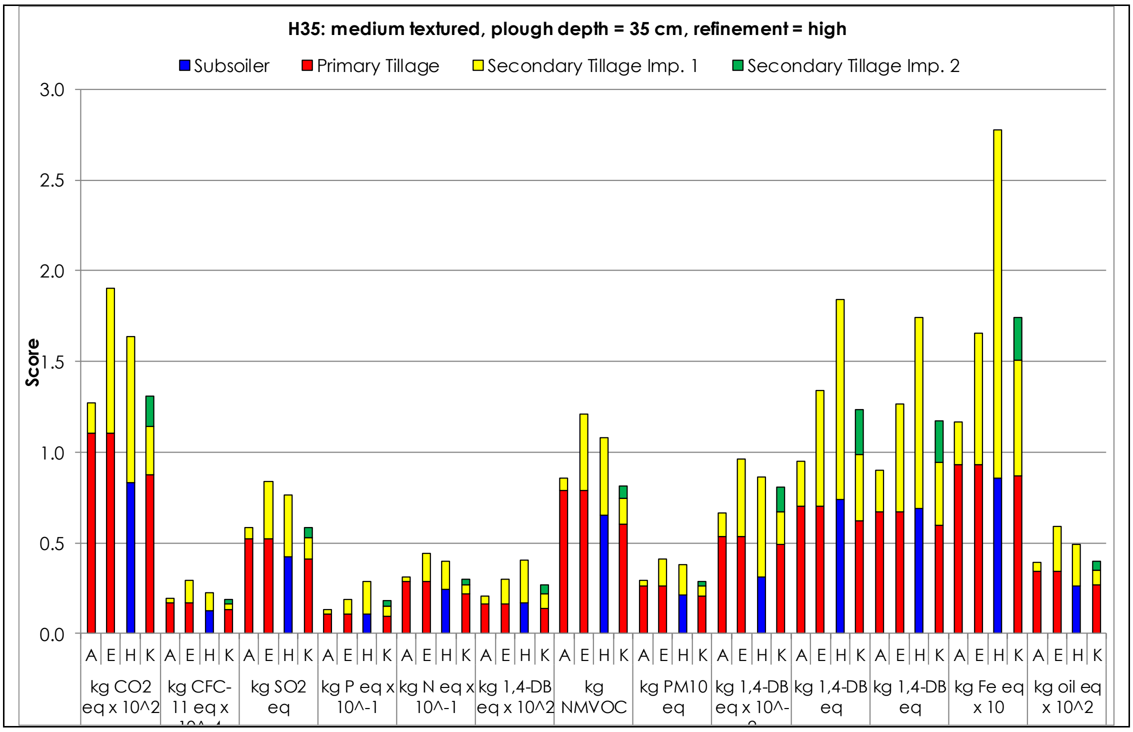
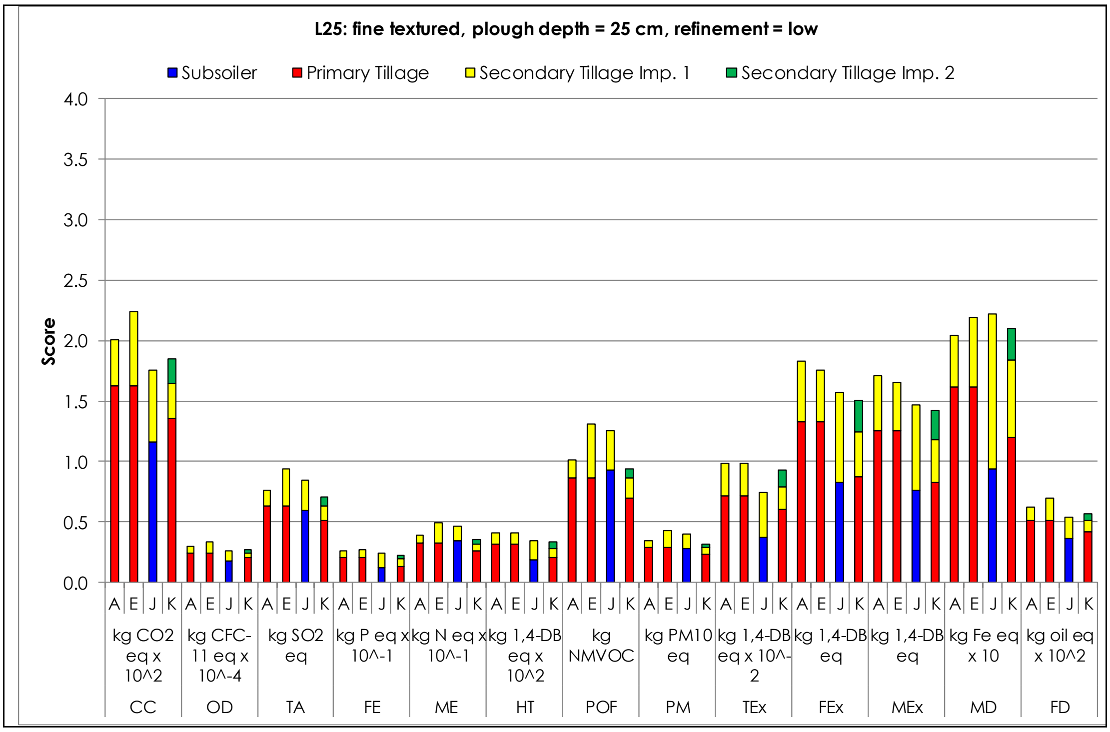


Figure 27b. Environmental results of the seedbed preparation on a medium textured soil with both ploughing depths (d_1 = 25 cm and d_2 = 35 cm) and low (L) or high (H) refinement during the secondary tillage.



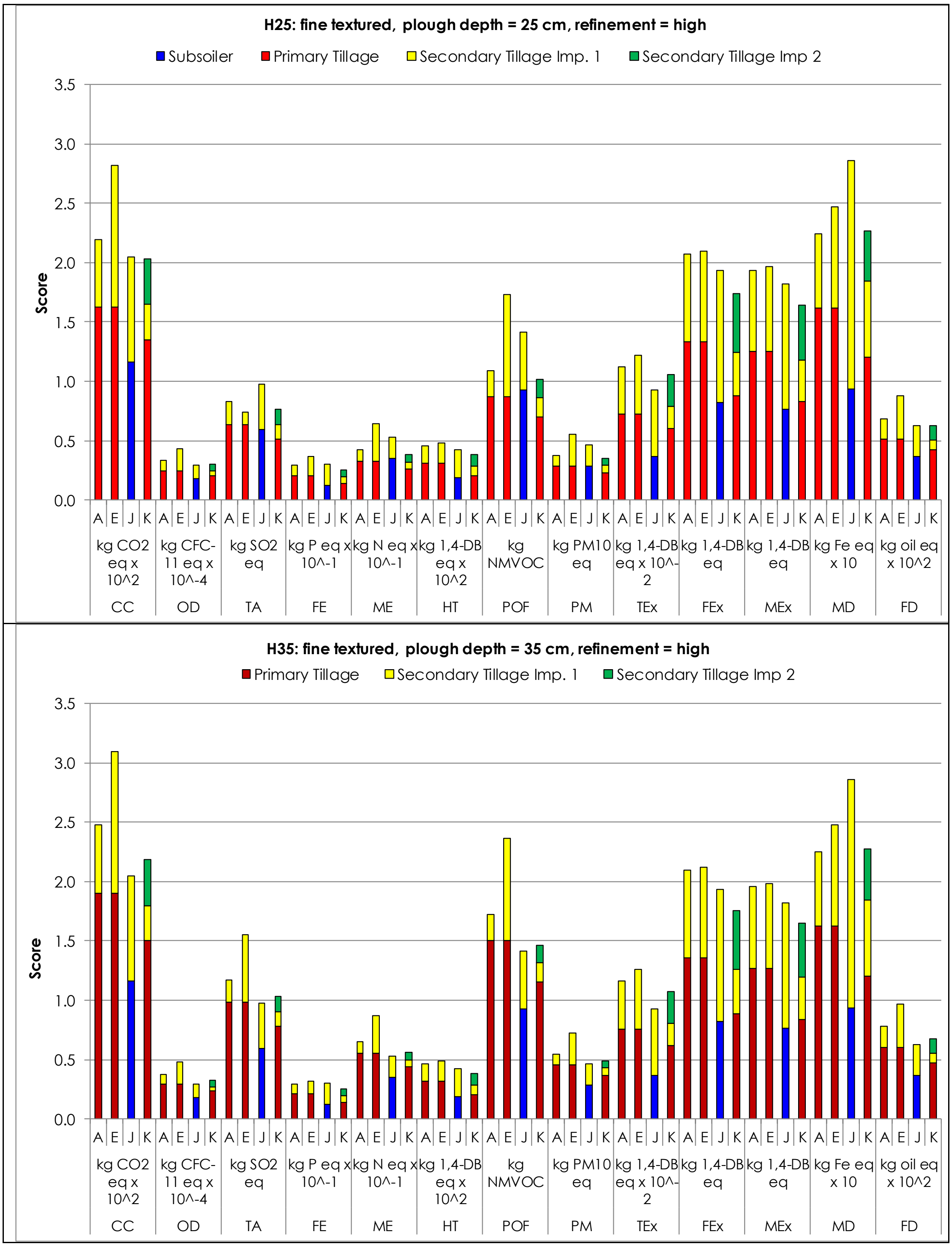


Figure 27c. Environmental results of the seedbed preparation on a clayey soil (fine textured) with both ploughing depths (d_1 = 25 cm and d_2 = 35 cm) and low (L) or high (H) refinement during the secondary tillage.

The goal of these evaluations is to compare the alternative solutions for seedbed preparation within the same soil texture, ploughing depth and refinement intensity. In fact, it is already qualitatively known that on sandy soils the environmental impact is commonly lower than on clayey soils, but with this study the difference in environmental impact among alternative solutions adoptable on the same soil and with the same soil features (ploughing depth and refinement intensity) have been quantified. Thus, observations and evaluations among the different implements can be made, mainly because work efficiency and tractor-implement coupling are essential for more sustainable soil tillage operations.

Although secondary soil tillage is energy-consuming, primary soil tillage has the highest contribution in all options and on all evaluated impact categories. The case with the slatted plough highlights the best environmental performances on all evaluated impact categories, due the lower implement mass and tractor fuel consumption along its use.

In all the evaluated cases, the options "C" (for the sandy soil), "E" and "H" (for the medium textured soil), and "E" and "J" (for the clayey soil) show the worst outcomes on all evaluated impact categories. For option "C" (disc plough and spring tine harrow) the reason is attributed to the higher impact gathered during disc ploughing respect to the mouldboard ploughing. In option "E", the rotary harrowing plays a prominent role on the seedbed preparation due to the related high fuel consumption and/or number of repetitions (respectively for the two cases). As regards option "H" (subsoiler and disc harrow), the most important reason for the high environmental impact is due to the number of repetitions of secondary soil tillage (i.e. 2 and 3), whereas for option "J" (mouldboard plough, spring tine harrow and rotary harrow) the reason is related mainly to the presence of two implements for the secondary tillage as well as to the high environmental impact related to ploughing. In support of this result, once more, the mass of implements and the more fuel consumed during their use (which is dependent on the absorbed engine power) are the major responsible for these less environmentally beneficial results. However, not in all the cases in which no ploughing was performed the environmental impact was the lowest. For a detailed analysis of the results, see attached *Paper 3*.

6.1.2 Fertilisers spreading (Paper 4)

An analysis of the fertilisers spreading operations with a complete vision on emissions to air, water and soil was performed thanks to LCA. From the study of *Paper 4* it was found that, although direct soil injection is the best solution for ammonia emission reduction, it also involves side effects.

In particular, the reduction of nitrogen emissions to air (ammonia volatilisation) caused an increase of emissions to soil and water (nitrate leaching and phosphate run-off) at the end of the cultivation season. The evaluated scenarios are reported in **Table 5**.

Table 5. Baseline scenario (BS) and alternative scenarios (AS) for the fertiliser spreading.

Option	Description	Timing of soil incorporation for organic fertilisers	Adopted fertilisers
BS	Incorporation	> 3 days after spreading	Pig slurry and urea
AS1	Fast soil incorporation	< 2 h after spreading	
AS2	Direct soil injection	During spreading	
AS3	Incorporation with straw collection	> 3 days after spreading	Pig slurry and urea + maize straw collection
AS4	Incorporation with digestate spreading	> 3 days after spreading	Digestate and urea
AS5	Mineral fertiliser - Urea for N	Not applicable	Urea and triple superphosphate
AS6	Mineral fertiliser - CAN for N		CAN and triple superphosphate

In **Table 6** are listed the studied environmental impact categories, while in **Figure 28** are reported the results of alternative scenarios for fertilisers spreading.

Table 6. Impact categories selected for the study from ILCD characterisation method.

Impact category	**Symbol**	**Unit**
Climate Change	CC	kg CO_2 eq
Ozone Depletion	OD	kg CFC-11 eq
Human Toxicity with carcinogenic effect	HTc	CTUh
Human Toxicity without carcinogenic effect	HTnc	CTUh
Particulate Matter Formation	PM	kg PM2.5 eq
Photochemical Oxidant Formation	POF	kg NMVOC eq
Terrestrial Acidification	TA	molc H+ eq
Terrestrial Eutrophication	TE	molc N eq
Freshwater Eutrophication	FE	kg P eq
Marine Eutrophication	ME	kg N eq
Freshwater ecotoxicity	FEx	CTUe
Mineral, fossil and renewable resources depletion	MFRD	kg SB eq

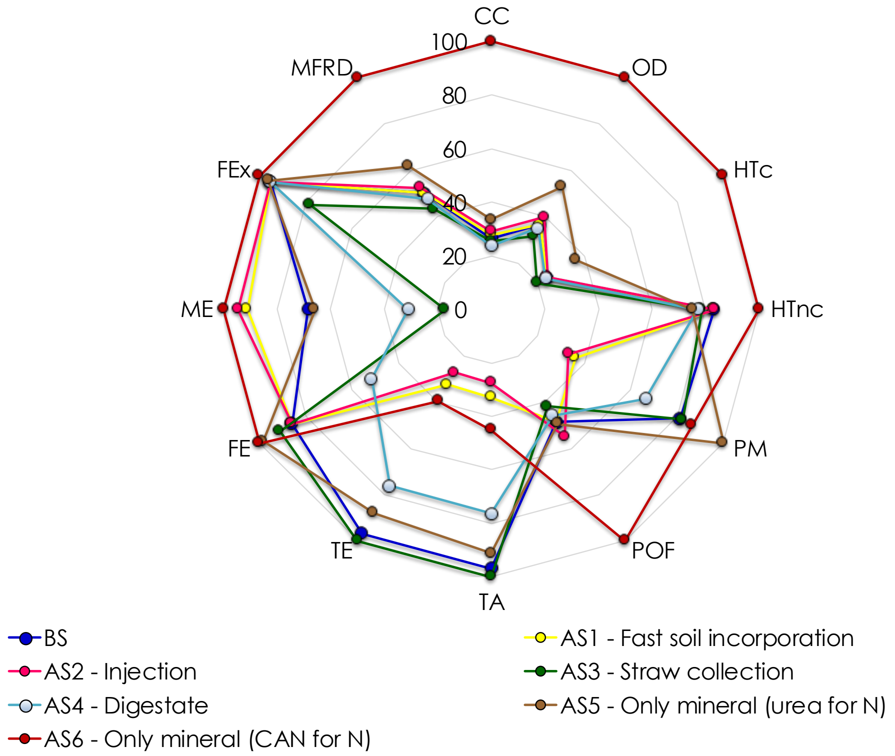


Figure 28. Environmental results for the compared strategies for organic and mineral fertilisers spreading.

In this case as well, the worst performing alternative is put at 100% of impact. Given the energy and primary sources for mineral fertilisers production, the worst solution for all environmental impacts except for TA and TE is the case in which only mineral fertilisers are spread (in particular AS6, where calcium ammonium nitrate - CAN - is used and not urea). For what regards the other cases, the environmental outcomes are close to each other for the impact categories affected by the field distribution, but are considerably different when considering the categories affected by the fertilisers emissions. In more details, the solutions that are commonly suggested for reducing ammonia emissions to air (AS1 and AS2) show low environmental impacts on TA and TE (about -66% and -72% respect to AS5), but high values for FE, ME and FEx, which are the categories on which the emissions to water play the main role. On the opposite, BS shows the most negative results on TA and TE, due to the late incorporation time.

A trade-off among the alternative fertiliser spreading techniques must be found, due to the not univocal results on all the evaluated impact categories. Having seen these environmental outcomes, it is important to evaluate, with a wider assessment, the sensitivity of the local systems in order to evaluate if is better to pursue a reduction of ammonia emissions or a reduction of nitrates and phosphates in water.

6.2 Data collection and prediction model: case study for enhancing mechanised field operations and LCA *(Paper 5)*

Considering the results gathered from the analysis of the field trials performed in Umeå (Sweden), in **Table 7** are shown data about the working time of the studied operations.

Table 7. Working time distribution (h) and Effective Field Capacity (EFC; ha/h) in the studied operations.

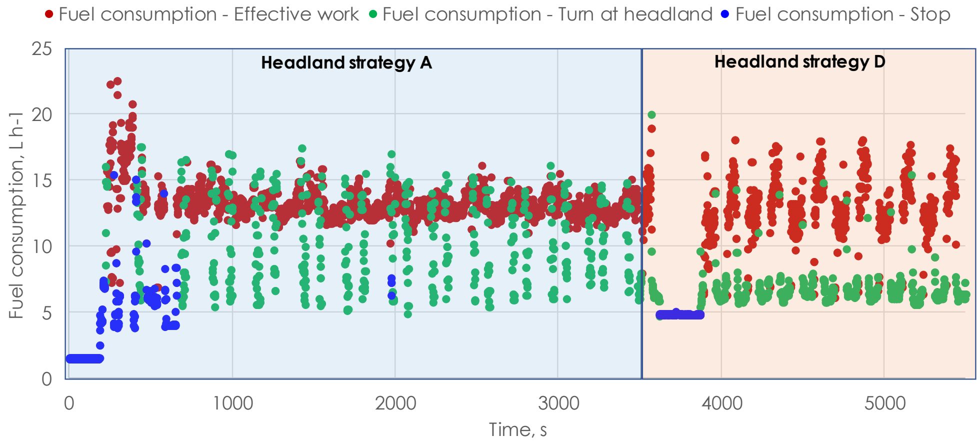
Operation	Effective work (h)	Turns (h)	Stops (h)	Transfer (h)	Total time (h)	EFC (ha/h)
Ploughing	1.93	0.62	0.39	1.46	4.40	0.28
Harrowing, rotary	1.77	0.82	0.36	1.40	4.35	0.39
Harrowing, spike	2.10	0.31	1.00 [1]	0.70 [2]	3.74	1.12
Sowing	0.69	0.16	0.35 [3]	1.19	2.39	1.76
Rolling	0.29	0.03	0.09	0.53 [2]	0.67	4.15

[1] Includes time to couple tractor-implement on field (implement already on field) and to change the work layout of the implement (i.e. change of working depth between two field parts).

[2] The spike harrow was already on the headlands of the field, therefore only the way back to factory is measured. Thus, the total time for transfer has been estimated doubling the value for the way forth and back.

[3] Includes time to refill the hopper with seed.

For every operation, the GPS signal allowed identifying the position on field of the tractor and attributing to each position the related value of fuel consumed and exhaust gas emitted, but also of the torque, engine speed, engine load, developed engine power, and other variables related to exhaust gases measurement (e.g., O_2 concentration and gas temperature). Finally, the outcomes were also grouped in the states of effective work, turns at headlands and stops, in order to identify the relation among engine variables and working state. This is illustrated, for all the studied operations, in **Figure 29a-e** with regard to the distinction in working states over time of the fuel consumption (dm^3/h; $L\ h^{-1}$).



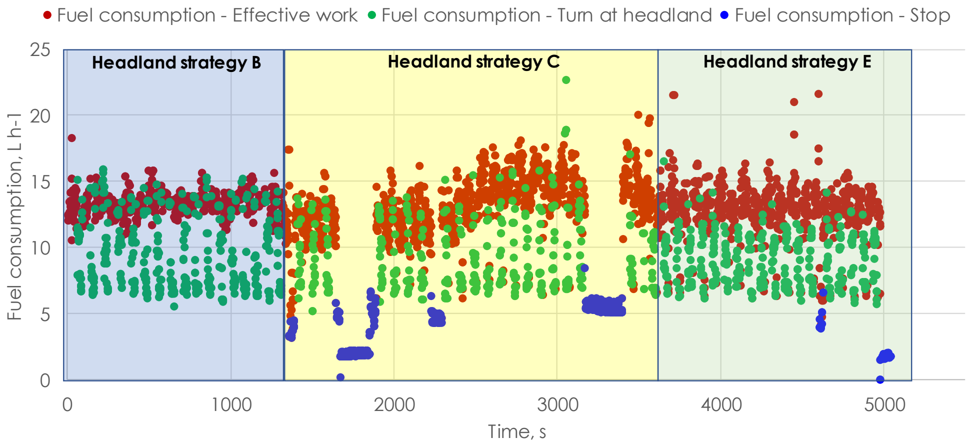


Figure 29a. Trend along time of the measured fuel consumption for the rotary harrowing with the strategies for the headlands named "A" and "D" on the top figure and "B", "C" and "E" on the bottom figure (see Figure 15 for the strategies code).

In Figure 29a the trend of fuel consumption over time is shown. In this case, rotary harrowing is shown, and the coloured boxes identify the field sections with different headland strategies. All strategies except for "C" and "D" show values in a restrained range of variation for what regards the effective work state (range between 12-16 dm^3/h in "A", "B" and "E"). In "C", the range is 10-18 dm^3/h, which is expected, since it is the case when the fieldwork engine speed was varied along the effective work state. In strategy "D", the variation ranges between 10-17 dm^3/h during the effective work because, also in this case, a change in gear was studied. During turns, the range is within 5-7 dm^3/h, due to the more homogeneous headland features respect to other strategies.

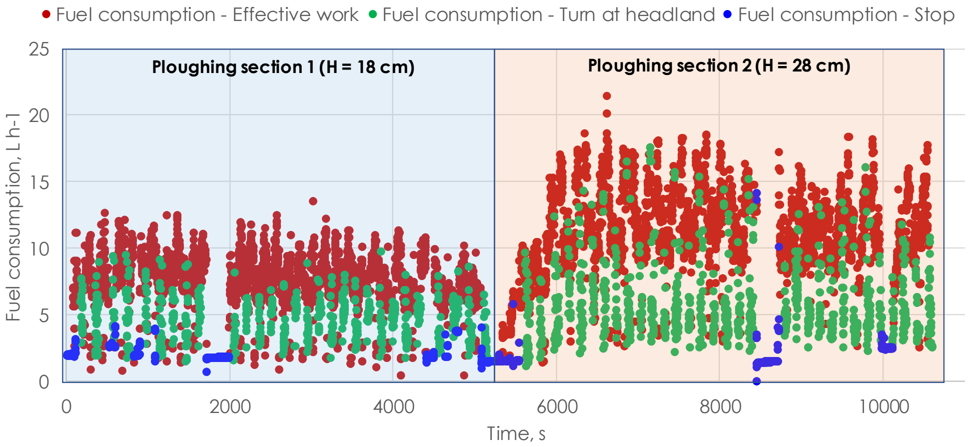


Figure 29b. Trend along time of the measured fuel consumption for the ploughing.

In Figure 29b is reported ploughing. In this case, a huge amount of data is available and a wider variability in the fuel consumption can be highlighted. The most interesting outcome is related to the different fuel consumption in the effective work depending on the ploughing depth (H_1 = 18 cm; H_2 = 28 cm). In the first part (with H_1), fuel consumption mainly ranges between 5-12 dm^3/h, while in the second (with H_2) the range is 7-18 dm^3/h. Regarding the turns, a similar trend is observed in both cases (2-8 dm^3/h) given the same headland strategy adopted.

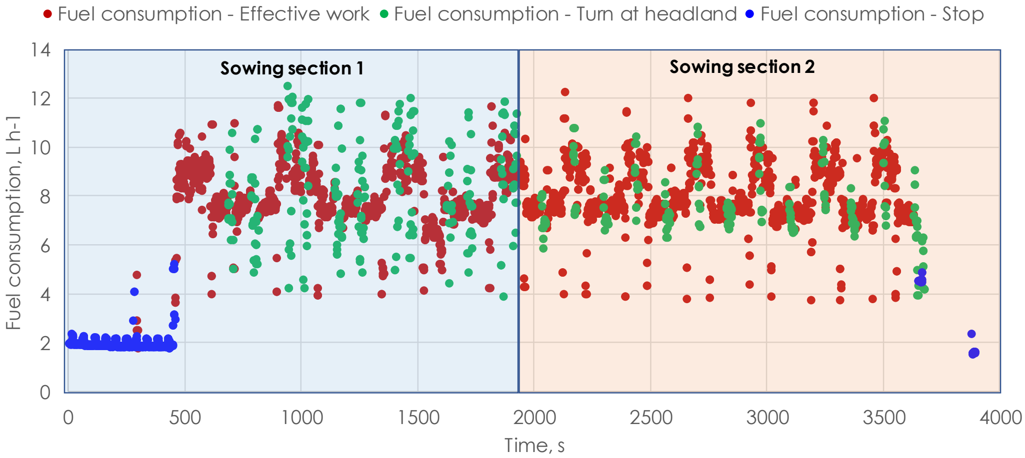
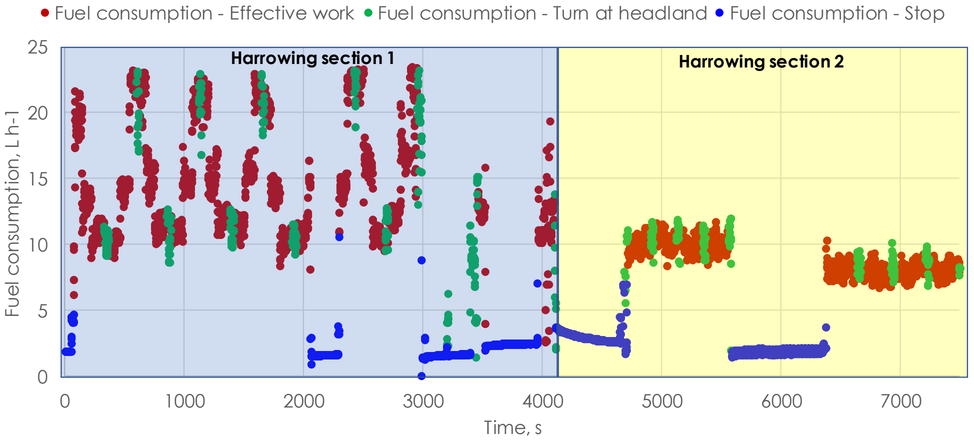


Figure 29c. Trend along time of the measured fuel consumption for the sowing.

Figure 29c refers to sowing. Section 1 for the "external" part of the field (worked with headland strategy "A") and section 2 for the "inner" part of the field (worked with headland strategy "E"). Both parts show a very similar trend referring to the effective work, while for turns a small difference emerges (Section 1 = 4-12 dm^3/h; Section 2 = 6-11 dm^3/h).



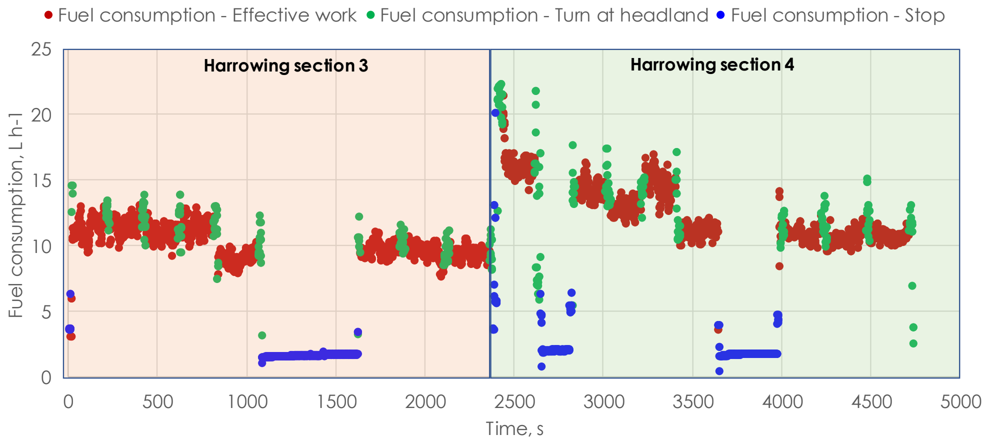


Figure 29d. Trend along time of the measured fuel consumption for the harrowing.

Harrowing has interesting outcomes: from Figure 29d emerges the effect of analysing the field by changing variables (engine speed and gear) on the same row (Section 1) or by keeping the same variables on a full section (Sections 2, 3 and 4). In particular, Section 1 was split in three parts on the length (see Figure 17) and this can be clearly recognised from the trend in fuel consumption of this part of the field. Sections 2, 3 and 4 show, instead, a consistent trend (7-10 dm^3/h, 9-11 dm^3/h and 11-14 dm^3/h, respectively).

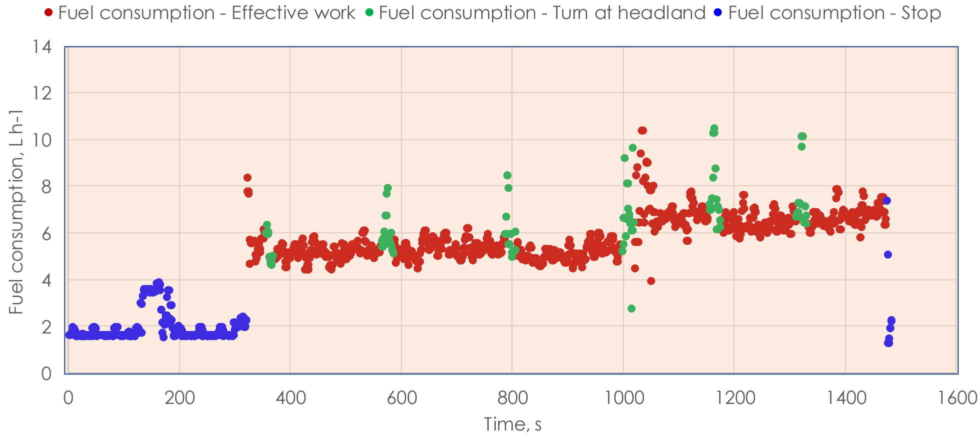
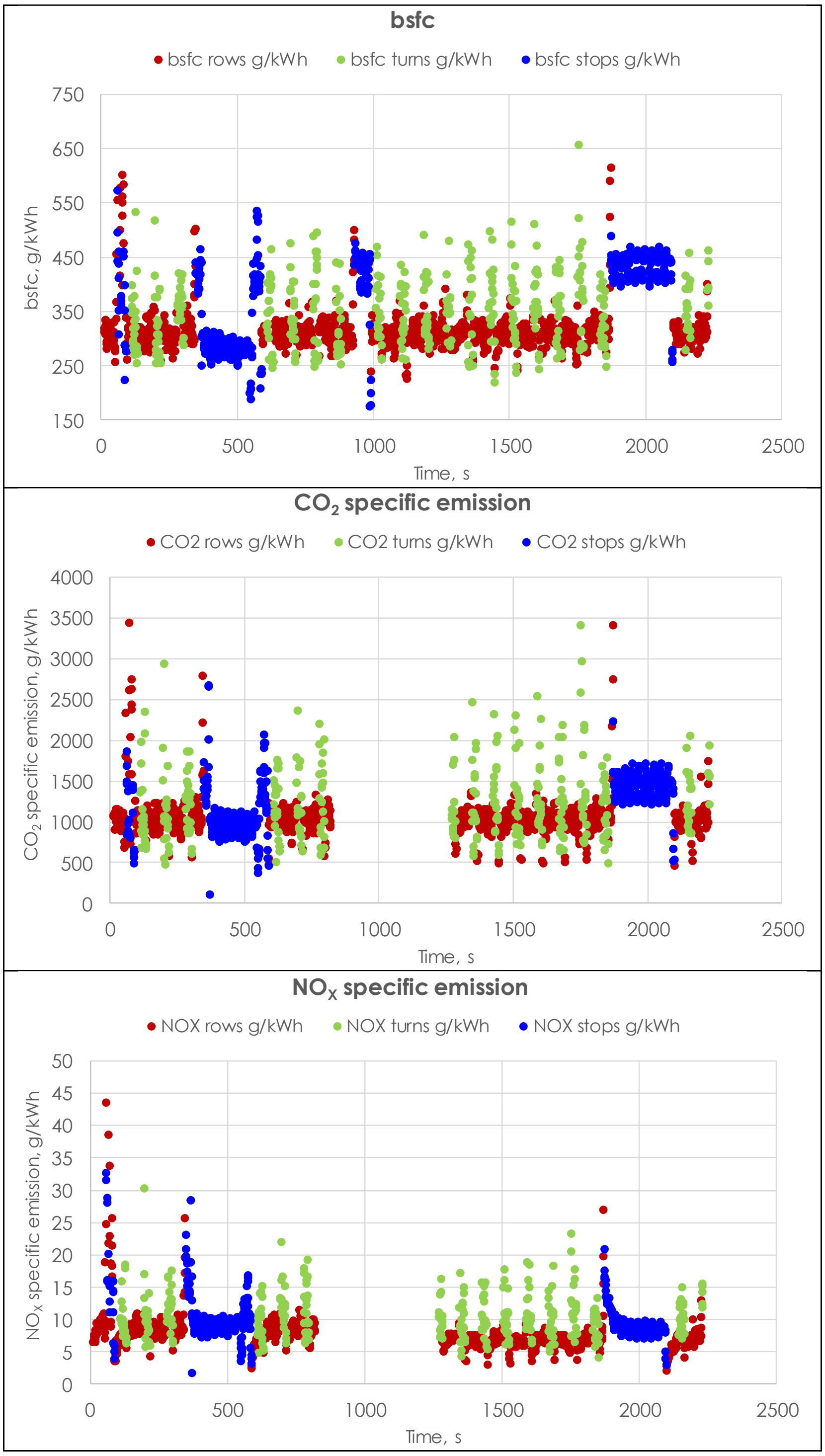


Figure 29e. Trend along time of the measured fuel consumption for the rolling.

The last operation, rolling, is reported in Figure 29e; the operation shows a very homogeneous result, mainly given the low engine power request and small variation in working features along the studied operation.

For every operation, the graphs shown in **Figure 30** (shown for rotary harrowing headland strategy “C”) were also gathered. They illustrate the trend over time of brake specific fuel consumption (bsfc; g/kWh) and specific engine exhaust emissions (EM; g/kWh) distinguished in effective work on rows, turns on headlands and stops.

The highest values are due to the turns and stops, while the effective work is characterised by lower specific fuel consumption and lower specific exhaust emissions: this entails a higher fuel efficiency for the effective work state respect to turns and stops.



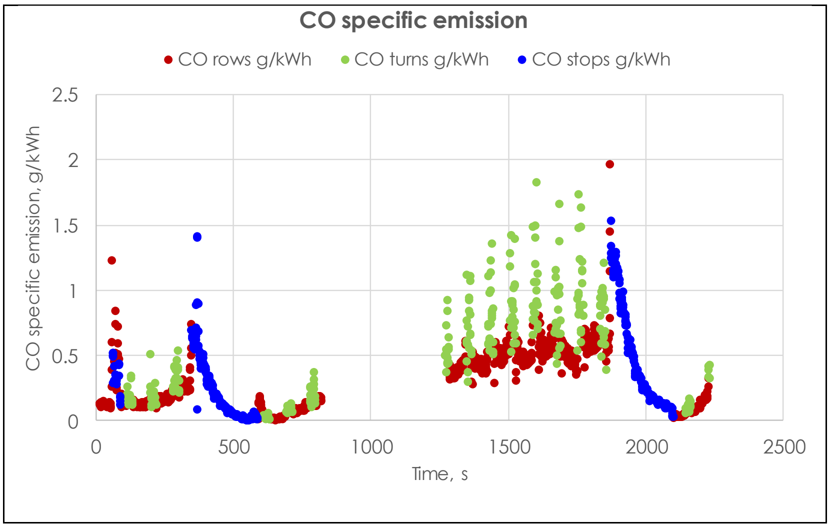
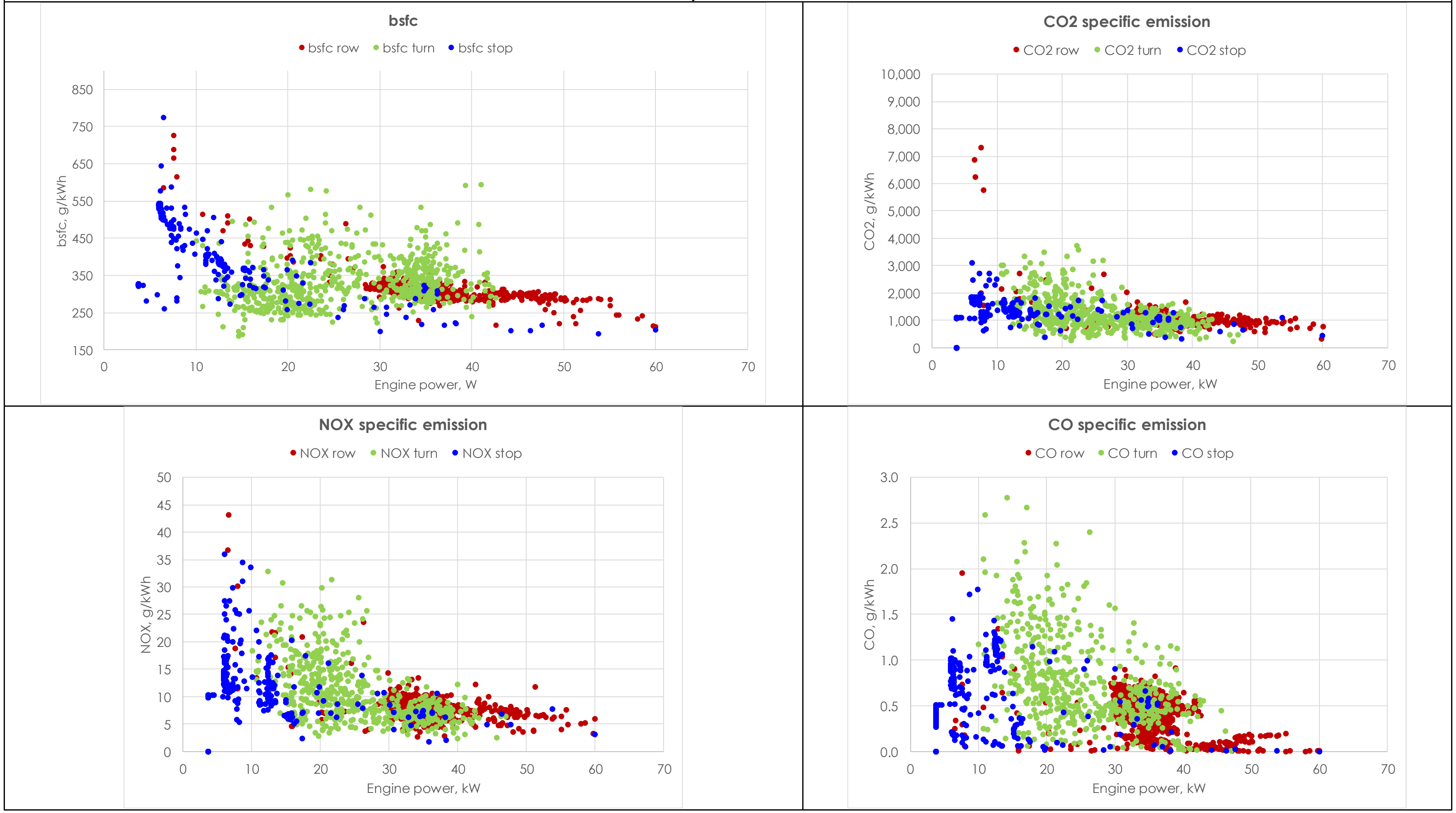


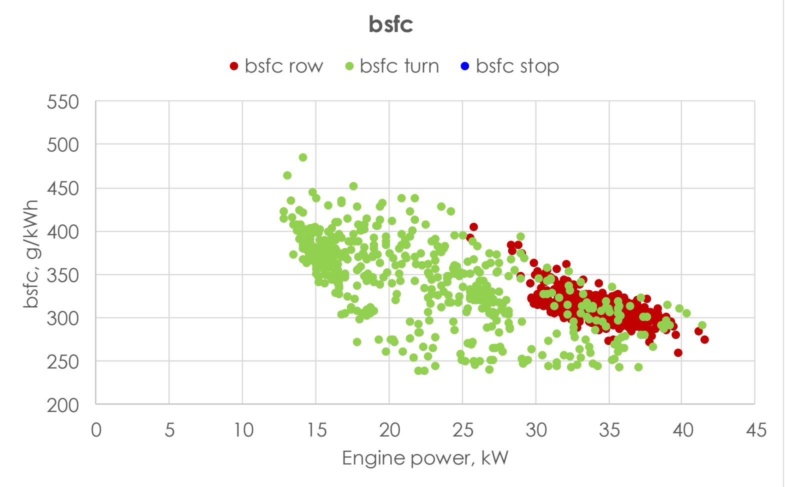
Figure 30. Trend over time (s) of brake specific fuel consumption (bsfc; g/kWh) and specific emissions of CO_2, NO_X and CO (g/kWh) for rotary harrowing within the headland strategy "C".

The same consideration can be made introducing engine power (kW) on the X-axis instead of time (s), allowing to identify the consumption and emission in relation with the developed engine power during the field operations, as illustrated in **Figure 31**. From these graphs, the concept of fuel efficiency emerges, since in almost all cases the values for bsfc and specific engine exhaust emissions show lower values during the rows of effective work. On the opposite, higher values are obtained during the turns at headlands and the stops, where the engine power is low. In more details, the highest values for almost all studied variables are shown when the developed engine power is low (< 20 kW, which means about < 25% of engine load). Values referred to stops are gathered either with the engine idling or when the tractor was stopping but the engine speed was kept constant with the effective work; this explains why some values refer to stops but are not associated with very low engine power. In most cases, however, the fuel consumption and exhaust gases emission linked with stops are the highest. Effective work on rows mostly shows the lowest values that are mainly obtained within high ranges of engine power; in few cases during some operations very high specific values are measured during the effective work on rows, but this occurs partially when the engine load is still low and partially when combustion features cause high CO specific emission. Considering NO_X, the low values obtained at high engine loads are due to the presence of the EGR system that permits to reduce these emissions by re-introducing part of the exhaust gases to avoid high temperature peaks.

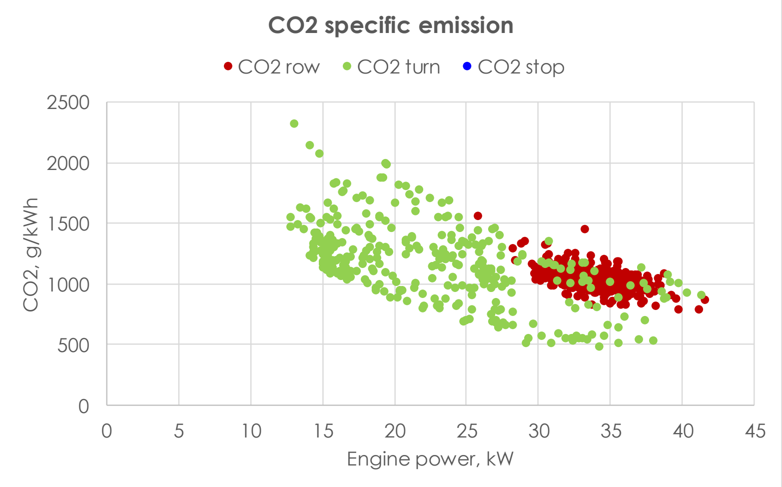
Rotary harrow “A”



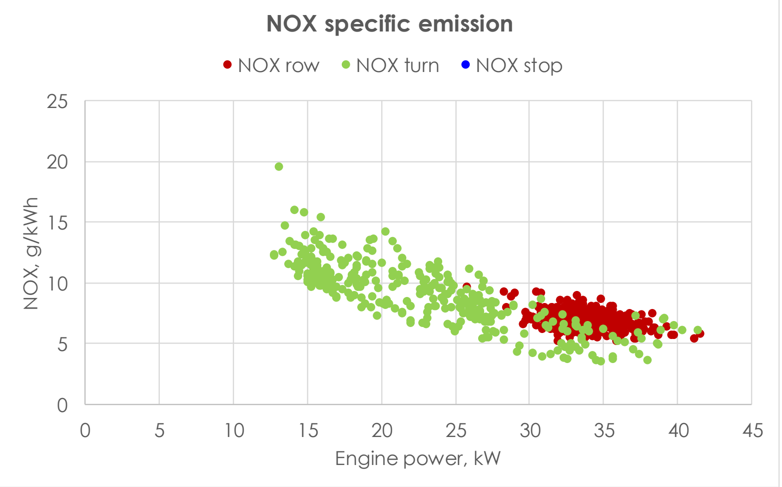
Rotary harrow “B”



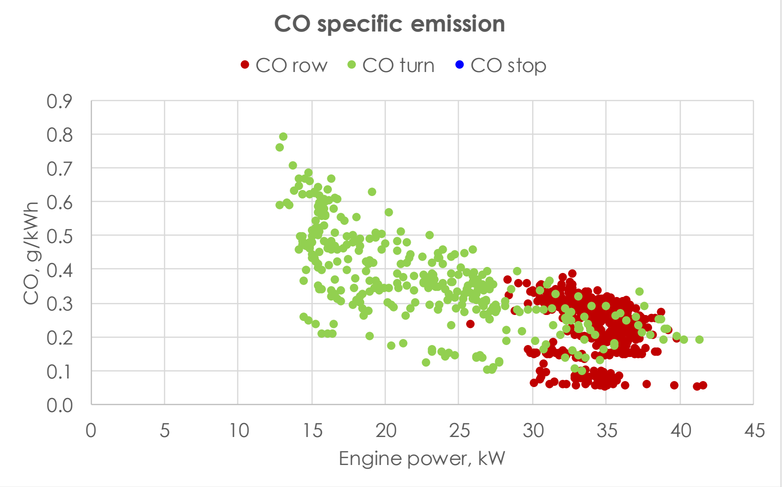
bsfc



CO2 specific emission

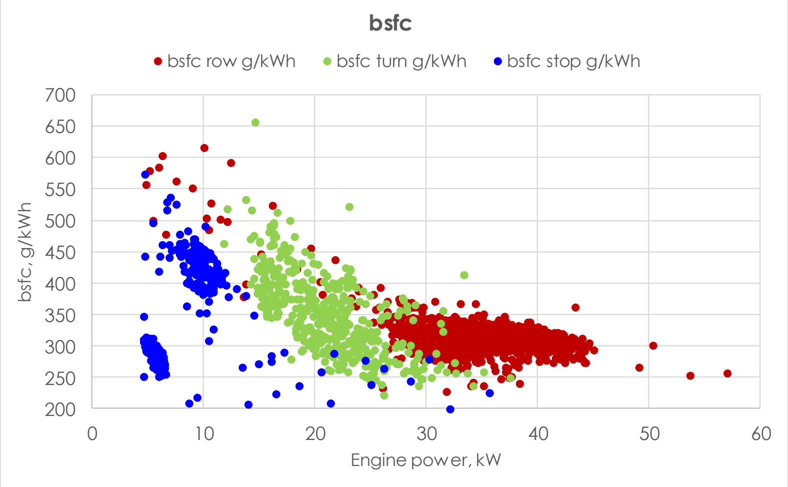


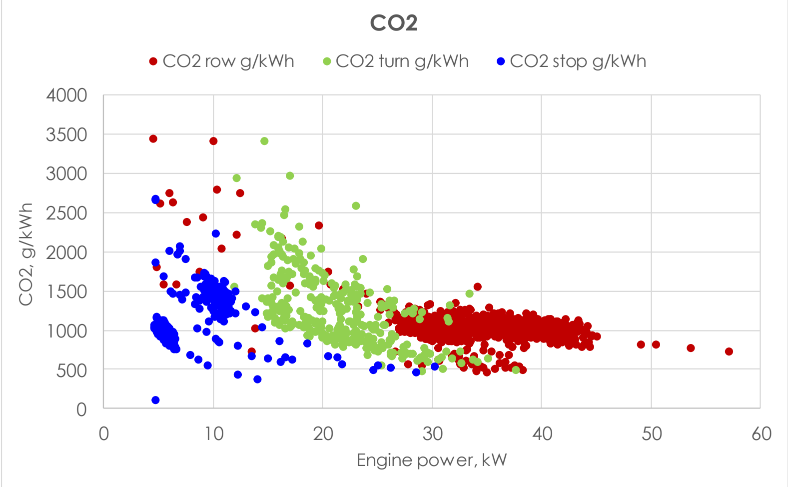
NOX specific emission

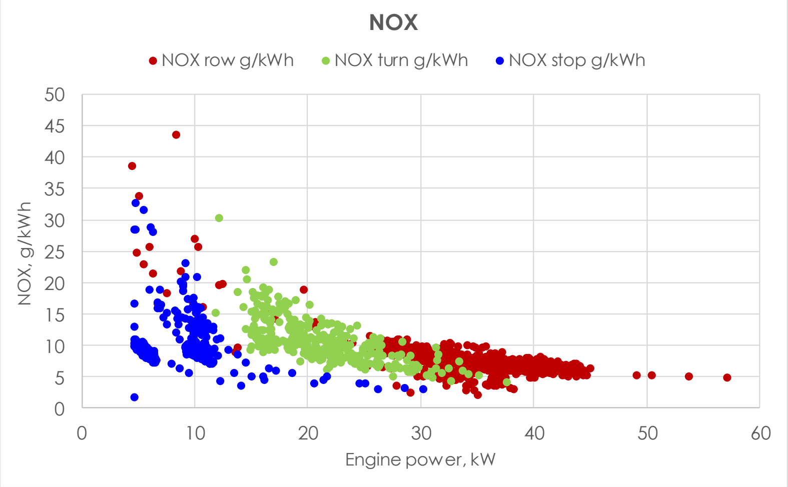


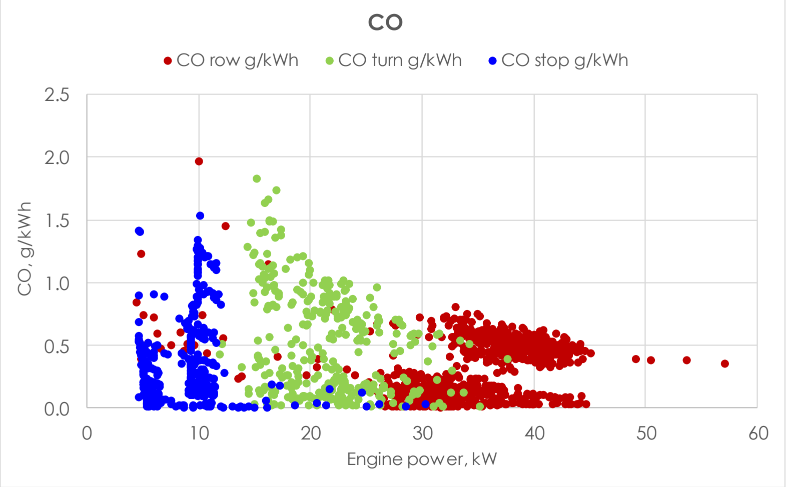
CO specific emission

Rotary harrow “C”

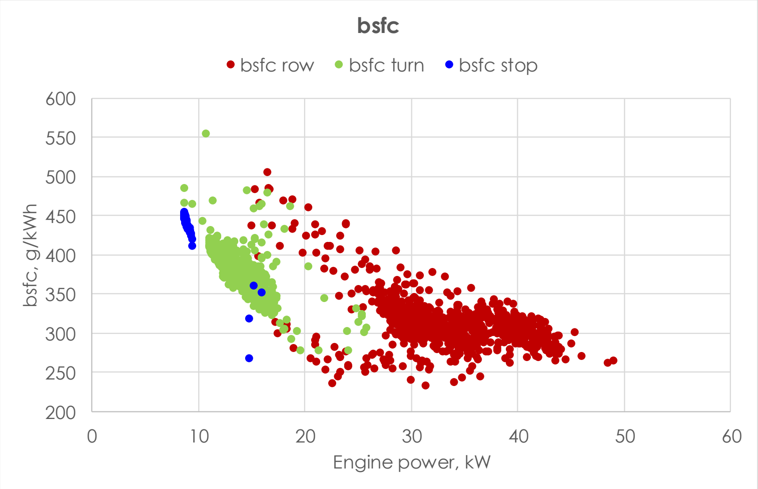


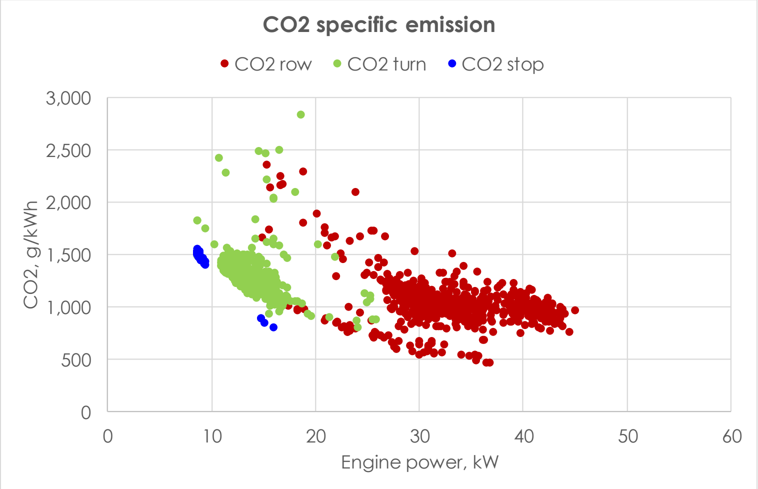


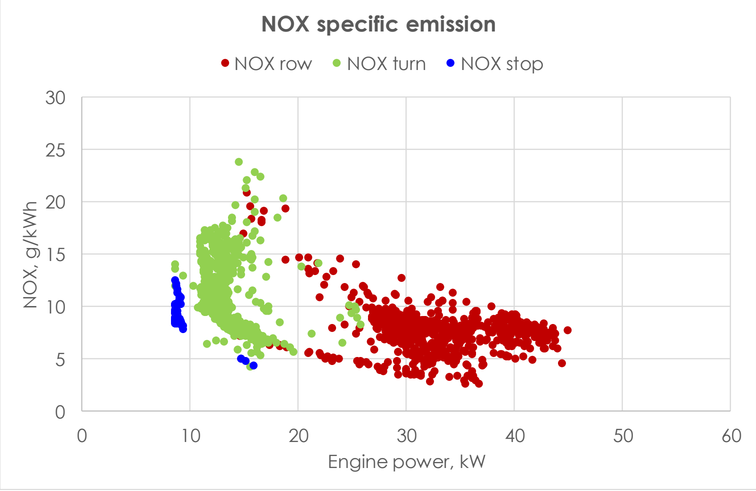


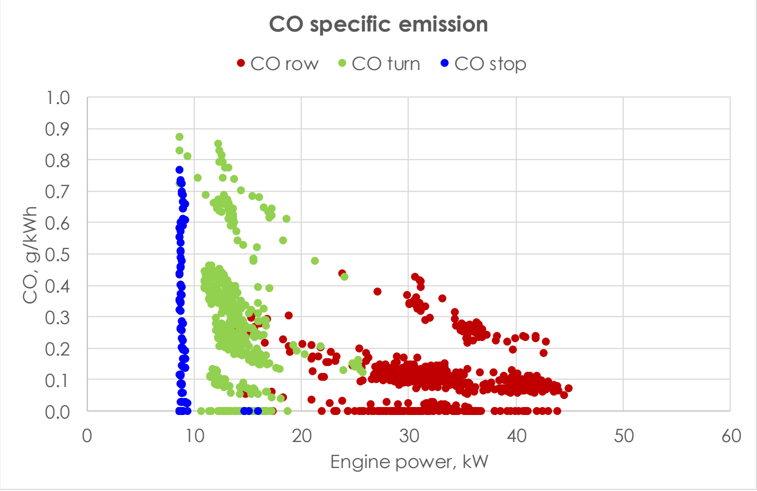


Rotary harrow “D”

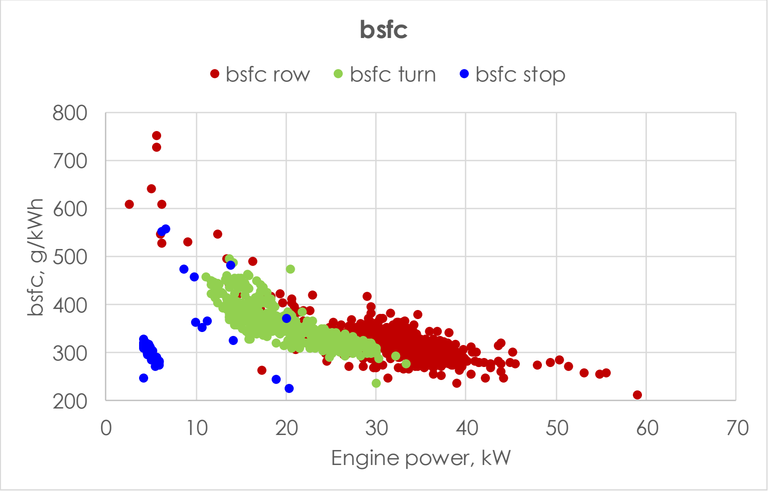


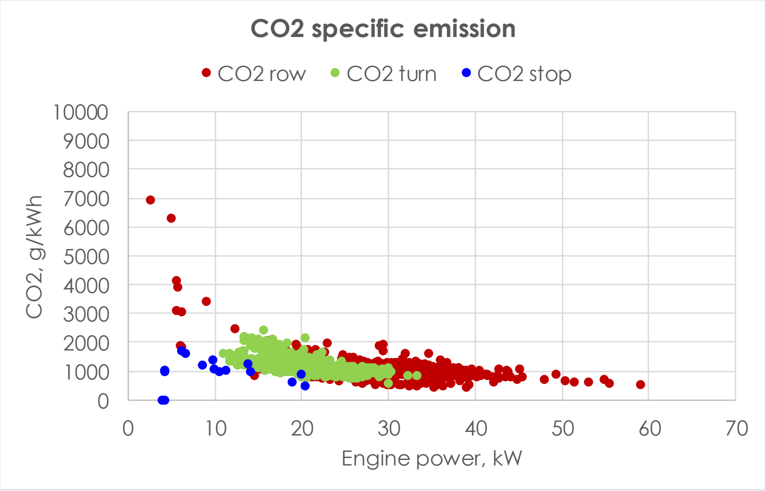


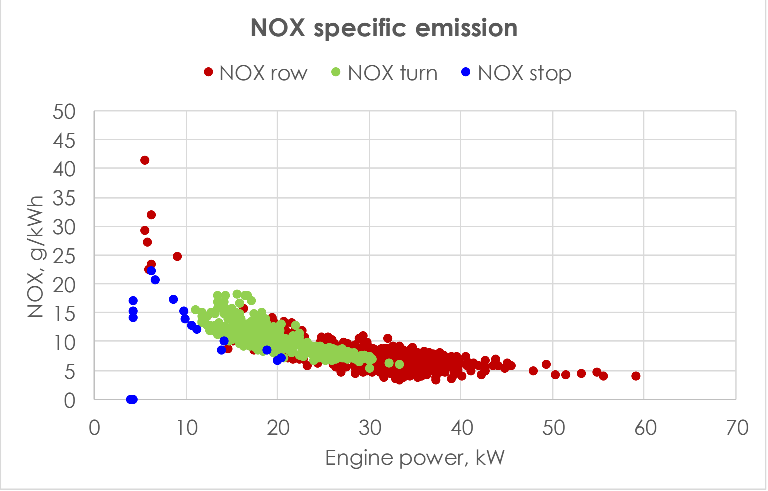


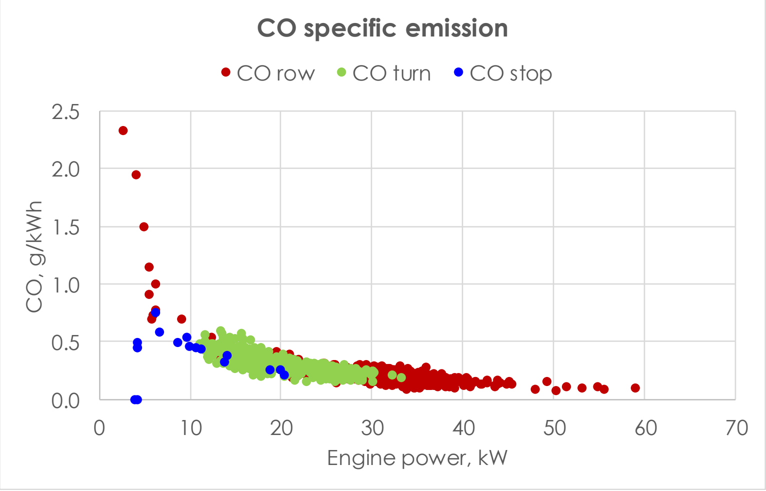


Rotary harrow “E”

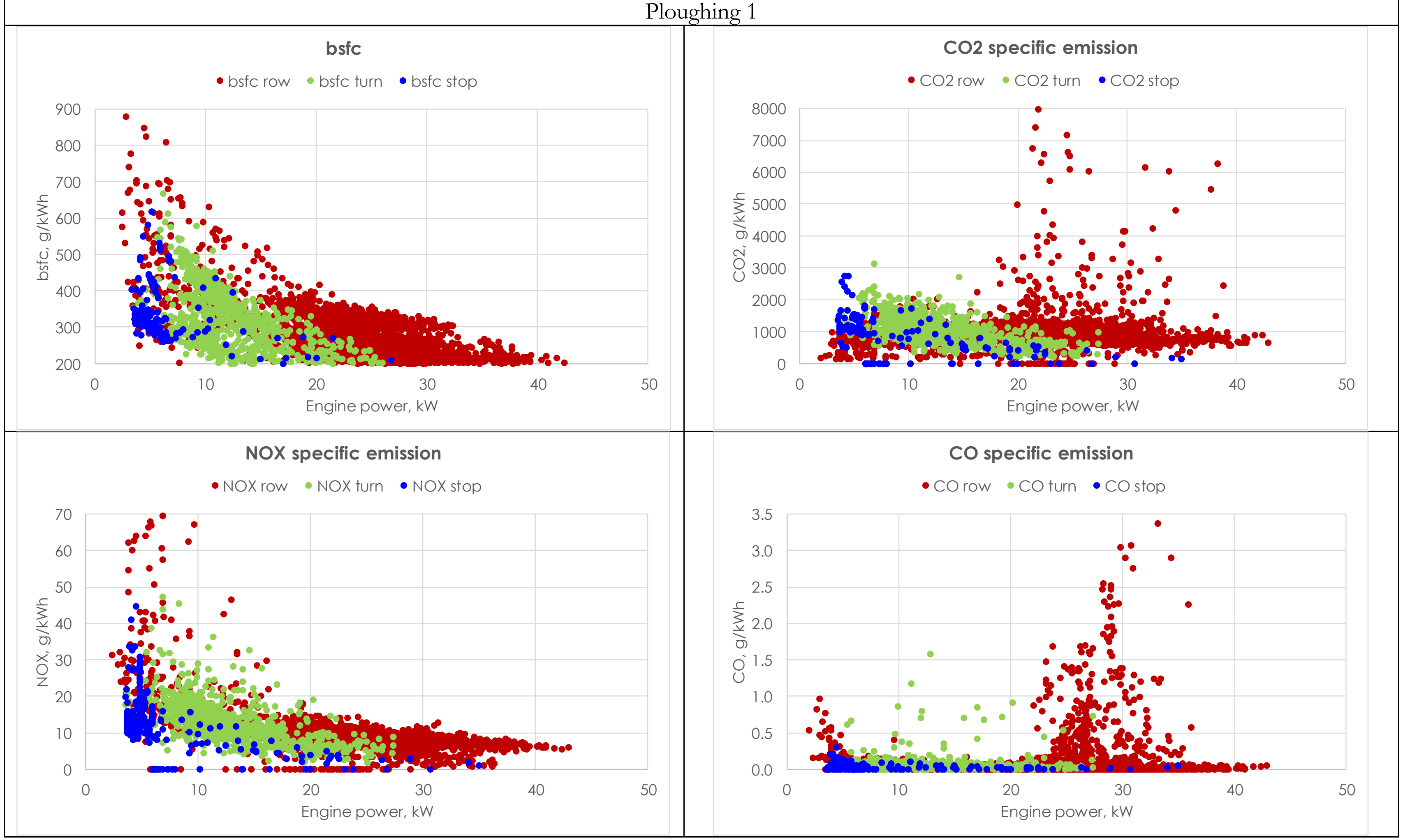

bsfc


CO2 specific emission

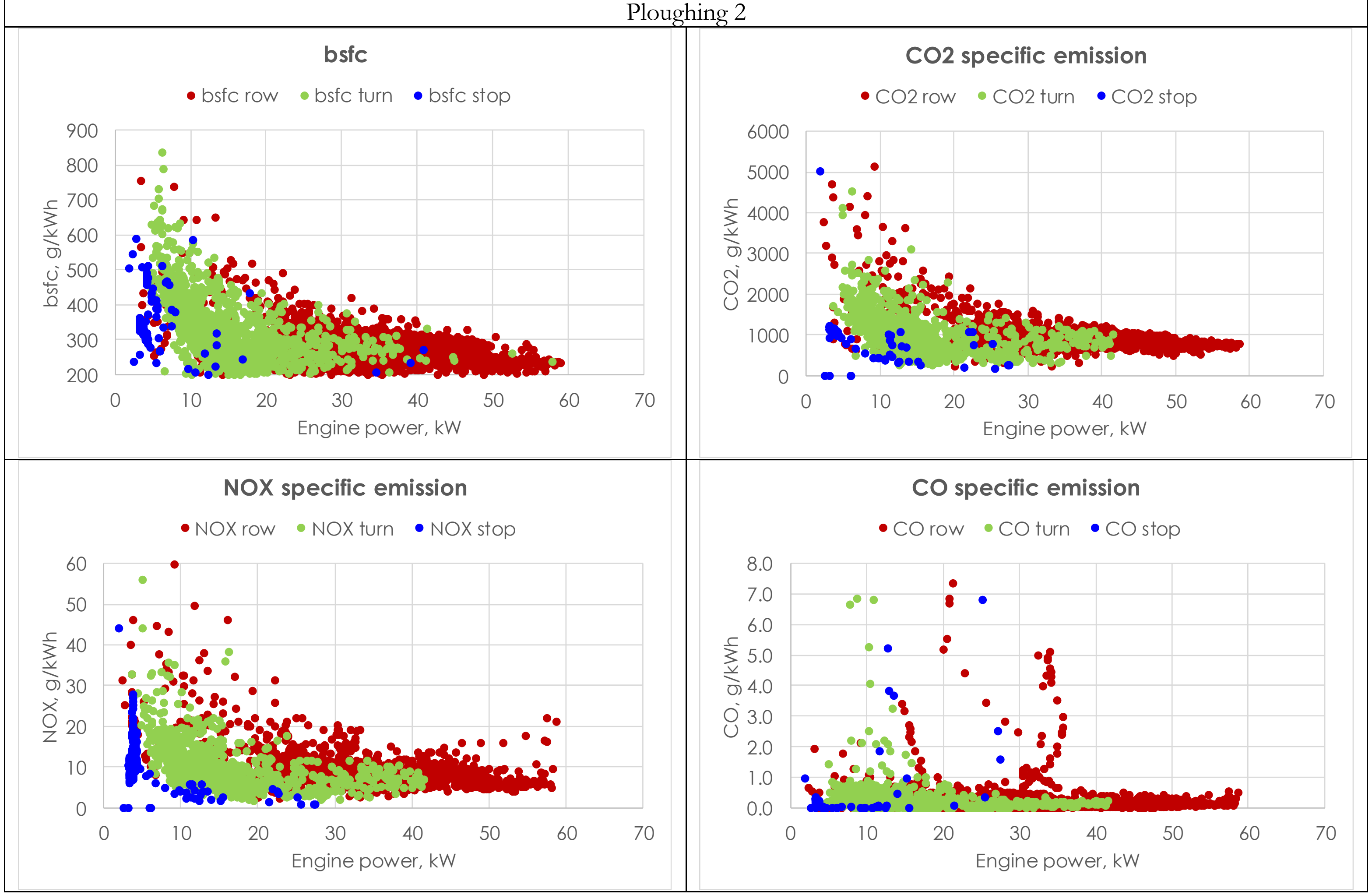

NOX specific emission


CO specific emission

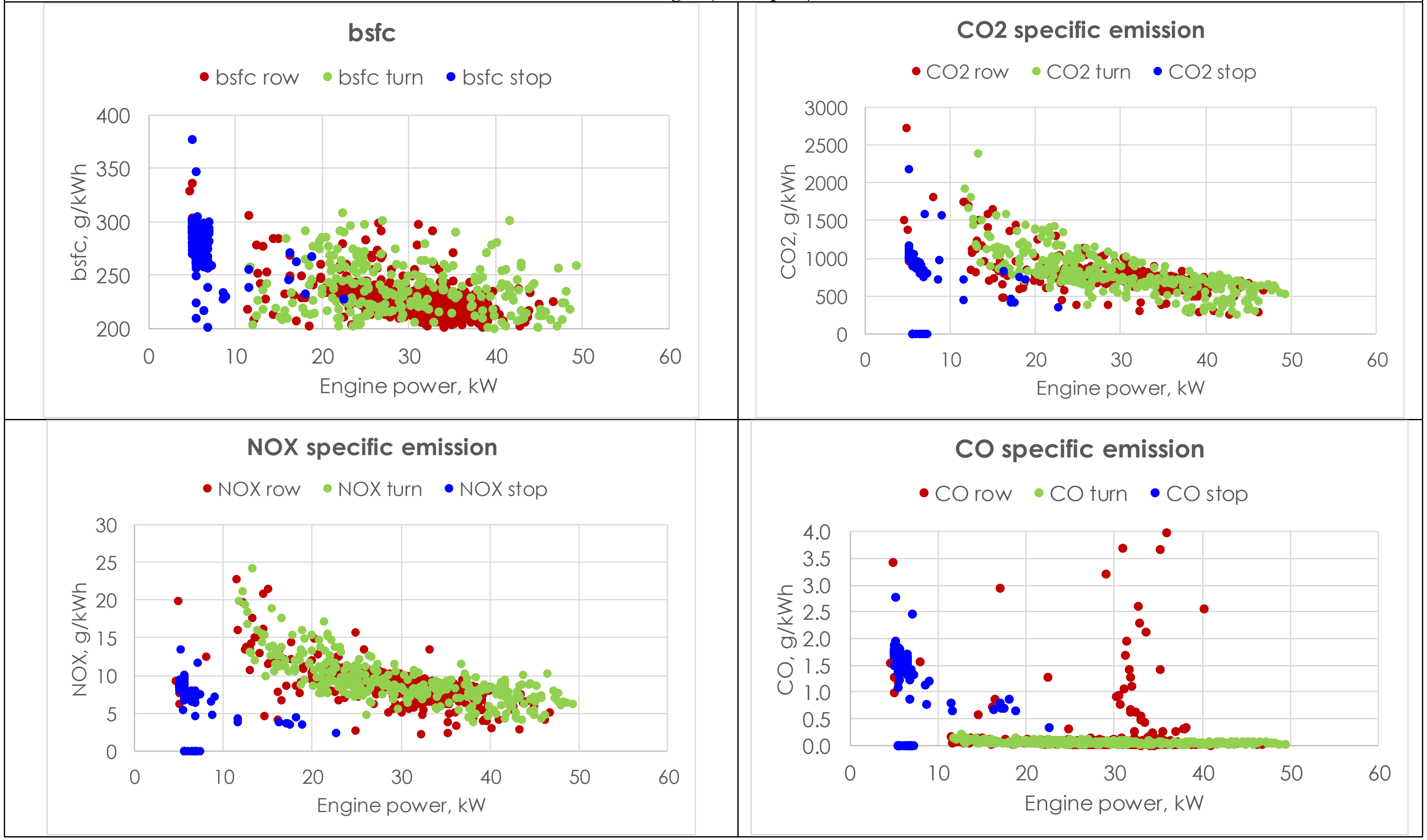
Ploughing 1


bsfc
bsfc row
bsfc turn
bsfc stop
bsfc, g/kWh
900
800
700
600
500
400
300
200
0
10
20
30
40
50
Engine power, kW
CO2 specific emission
CO2 row
CO2 turn
CO2 stop
CO2, g/kWh
8000
7000
6000
5000
4000
3000
2000
1000
0
0
10
20
30
40
50
Engine power, kW
NOX specific emission
NOX row
NOX turn
NOX stop
NOX, g/kWh
70
60
50
40
30
20
10
0
0
10
20
30
40
50
Engine power, kW
CO specific emission
CO row
CO turn
CO stop
CO, g/kWh
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
0
10
20
30
40
50
Engine power, kW

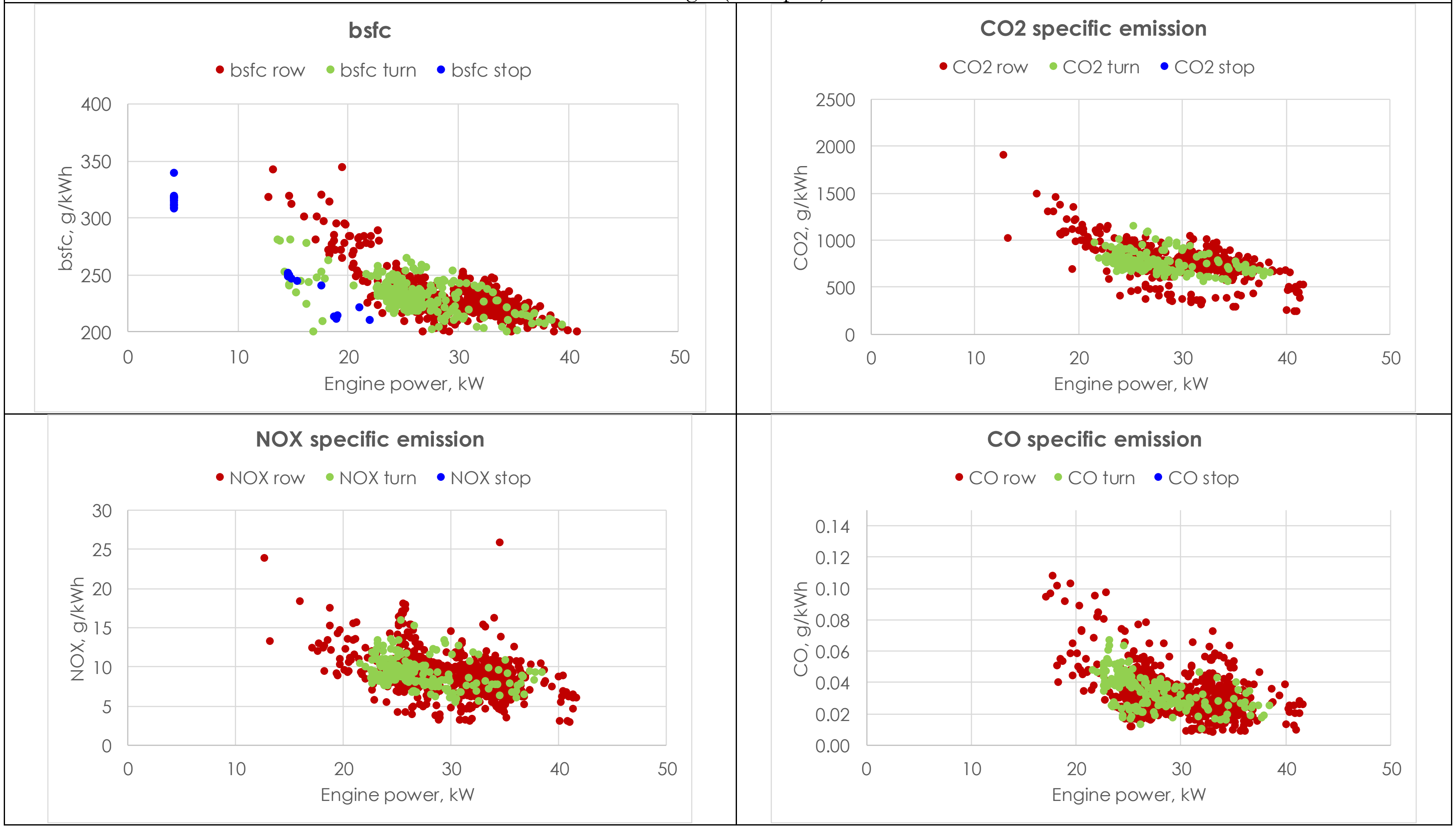
Ploughing 2


bsfc
● bsfc row ● bsfc turn ● bsfc stop
bsfc, g/kWh
900
800
700
600
500
400
300
200
0
10
20
30
40
50
60
70
Engine power, kW
CO2 specific emission
● CO2 row ● CO2 turn ● CO2 stop
CO2, g/kWh
6000
5000
4000
3000
2000
1000
0
0
10
20
30
40
50
60
70
Engine power, kW
NOX specific emission
● NOX row ● NOX turn ● NOX stop
NOX, g/kWh
60
50
40
30
20
10
0
0
10
20
30
40
50
60
70
Engine power, kW
CO specific emission
● CO row ● CO turn ● CO stop
CO, g/kWh
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
0
10
20
30
40
50
60
70
Engine power, kW

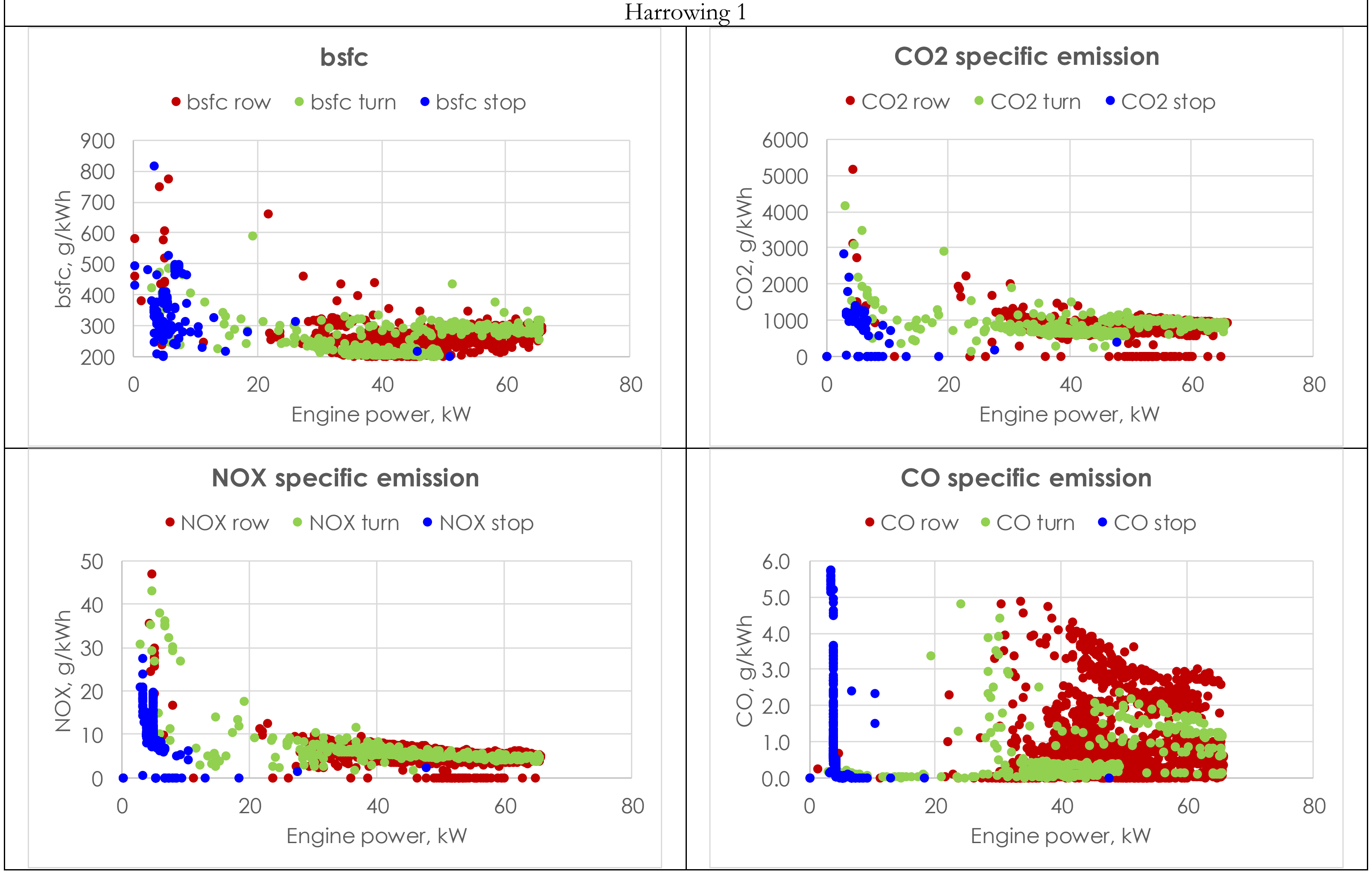
Sowing 1 (outer part)



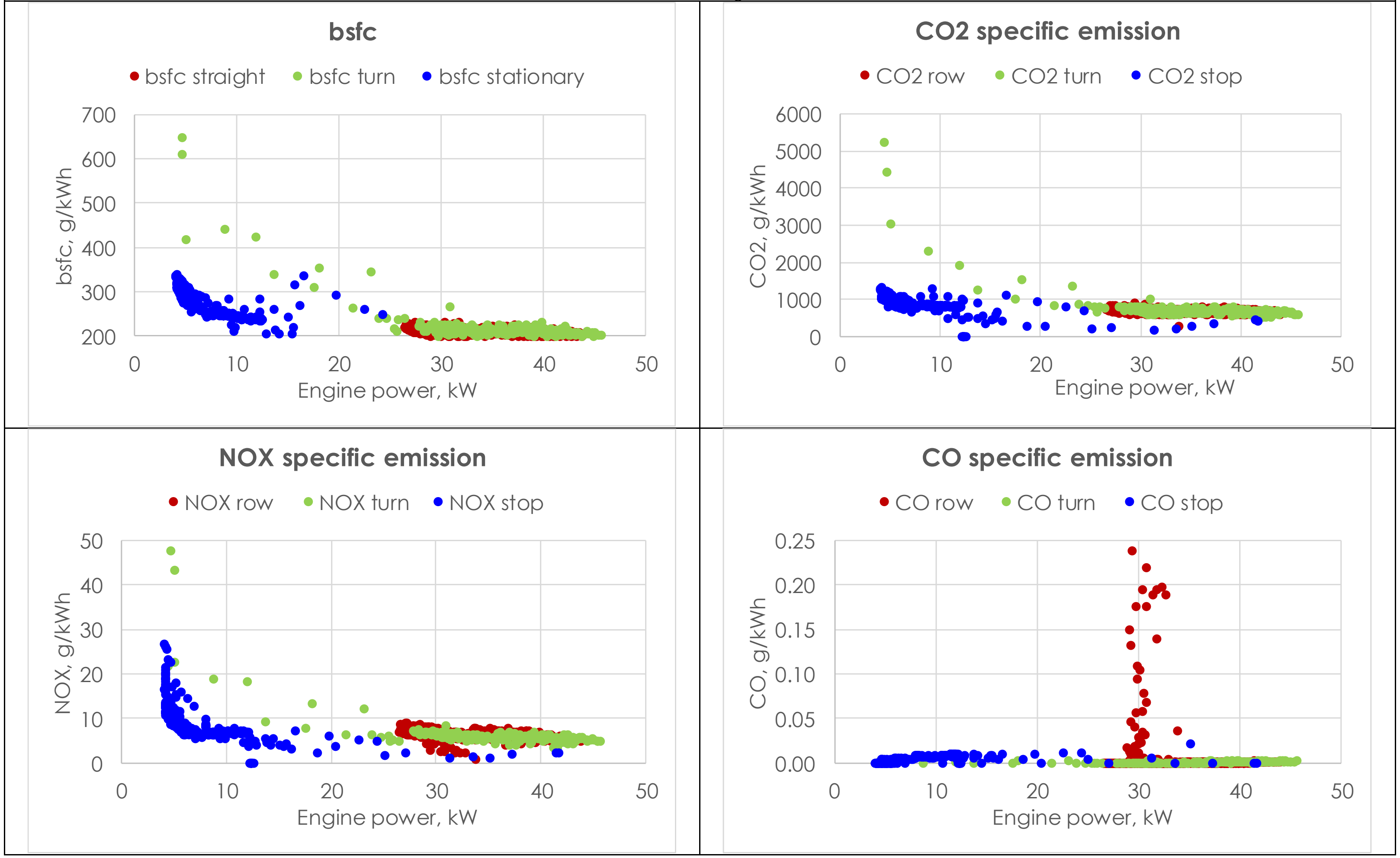
Sowing 2 (inner part)



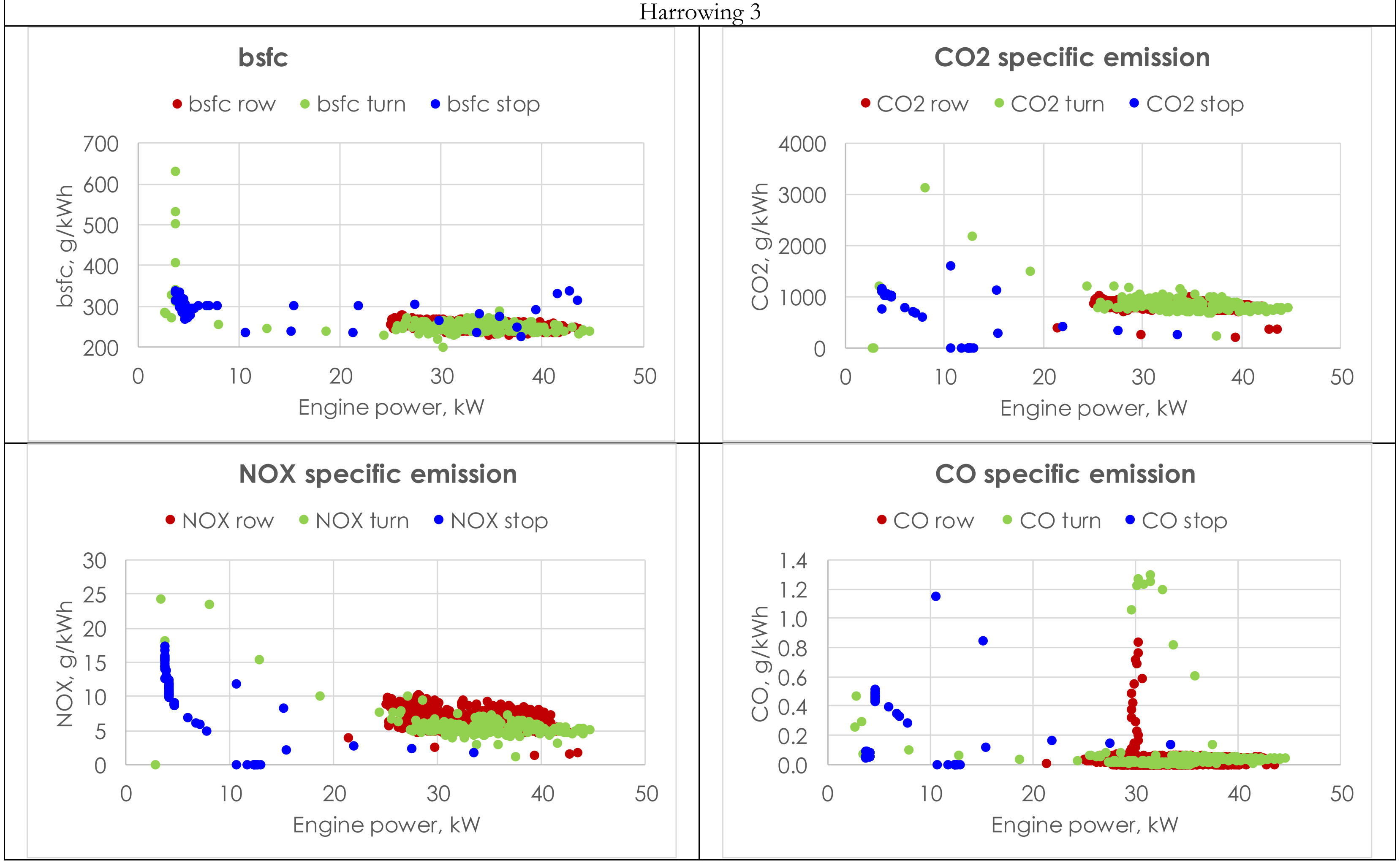
Harrowing 1


bsfc
bsfc row
bsfc turn
bsfc stop
bsfc, g/kWh
900
800
700
600
500
400
300
200
0
20
40
60
80
Engine power, kW
CO2 specific emission
CO2 row
CO2 turn
CO2 stop
CO2, g/kWh
6000
5000
4000
3000
2000
1000
0
0
20
40
60
80
Engine power, kW
NOX specific emission
NOX row
NOX turn
NOX stop
NOX, g/kWh
50
40
30
20
10
0
0
20
40
60
80
Engine power, kW
CO specific emission
CO row
CO turn
CO stop
CO, g/kWh
6.0
5.0
4.0
3.0
2.0
1.0
0.0
0
20
40
60
80
Engine power, kW

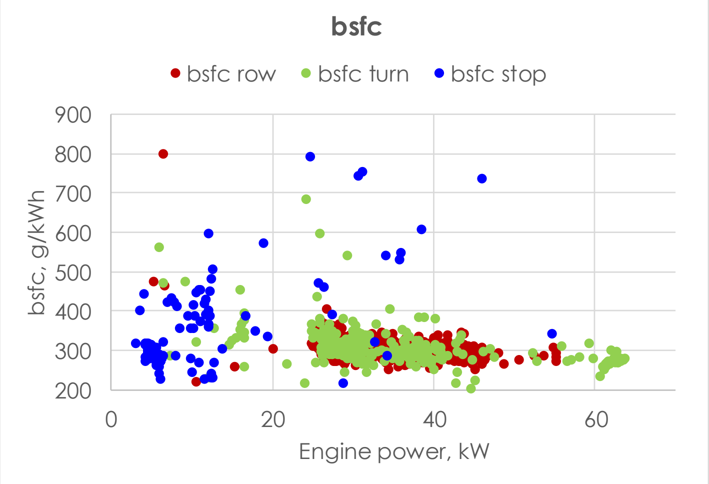
Harrowing 2

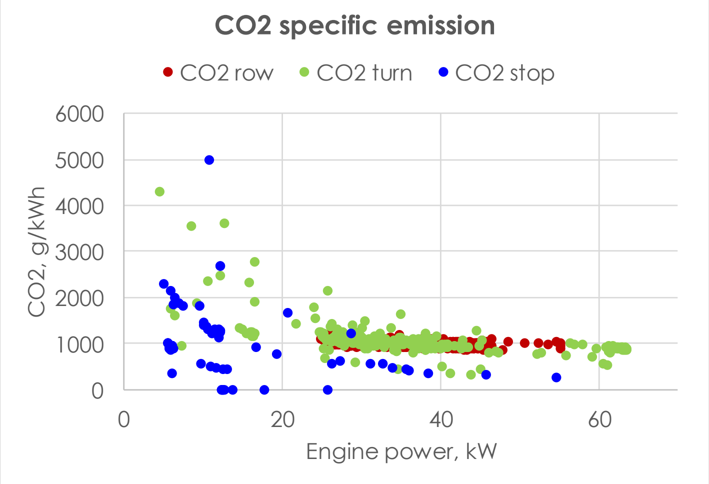

bsfc
bsfc straight
bsfc turn
bsfc stationary
bsfc, g/kWh
700
600
500
400
300
200
0
10
20
30
40
50
Engine power, kW
CO2 specific emission
CO2 row
CO2 turn
CO2 stop
CO2, g/kWh
6000
5000
4000
3000
2000
1000
0
0
10
20
30
40
50
Engine power, kW
NOX specific emission
NOX row
NOX turn
NOX stop
NOX, g/kWh
50
40
30
20
10
0
0
10
20
30
40
50
Engine power, kW
CO specific emission
CO row
CO turn
CO stop
CO, g/kWh
0.25
0.20
0.15
0.10
0.05
0.00
0
10
20
30
40
50
Engine power, kW

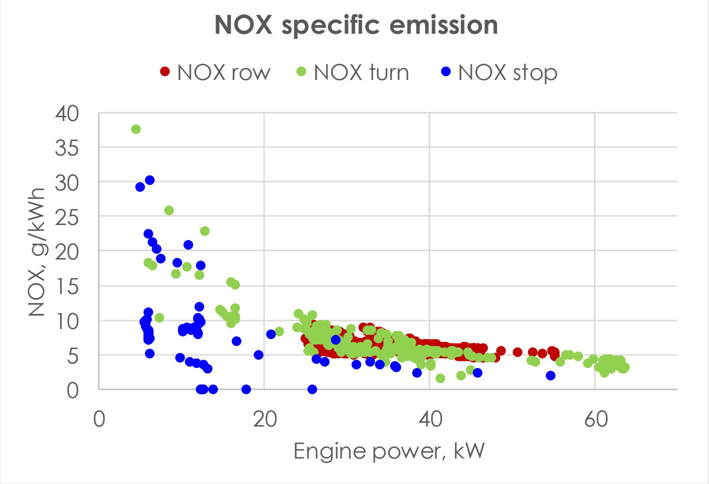
Harrowing 3

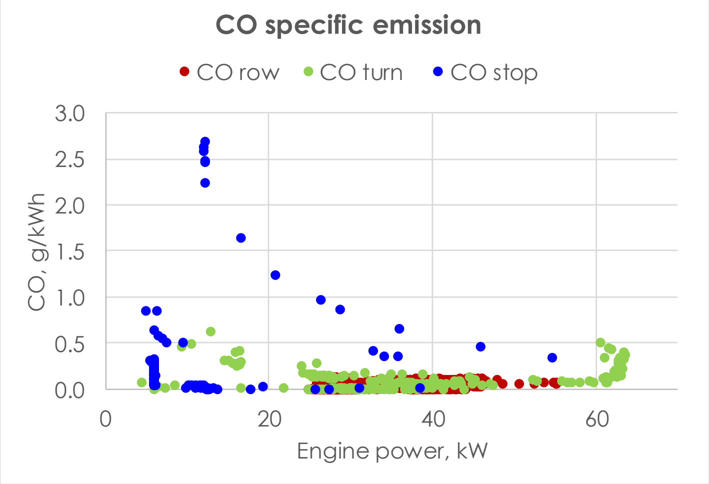

bsfc
bsfc row
bsfc turn
bsfc stop
bsfc, g/kWh
700
600
500
400
300
200
0
10
20
30
40
50
Engine power, kW
CO2 specific emission
CO2 row
CO2 turn
CO2 stop
CO2, g/kWh
4000
3000
2000
1000
0
0
10
20
30
40
50
Engine power, kW
NOX specific emission
NOX row
NOX turn
NOX stop
NOX, g/kWh
30
25
20
15
10
5
0
0
10
20
30
40
50
Engine power, kW
CO specific emission
CO row
CO turn
CO stop
CO, g/kWh
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0
10
20
30
40
50
Engine power, kW

Harrowing 4


bsfc


CO2 specific emission


NOX specific emission


CO specific emission

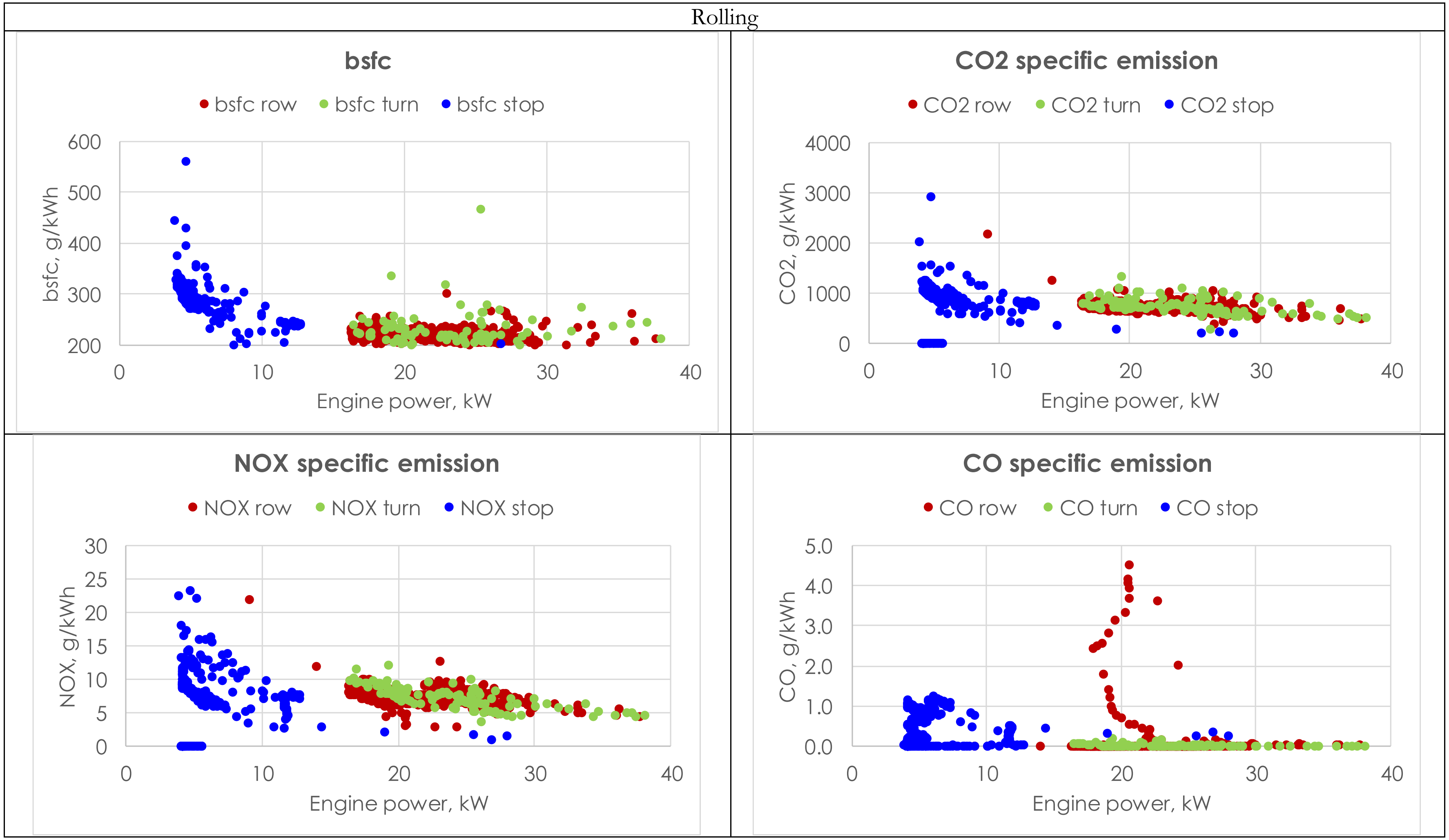


Figure 31. Trend over the developed engine power (W) of brake specific fuel consumption (bsfc; g/kWh) and specific emissions of CO_2, NO_X and CO (g/kWh) for all the studied operations.

For these results, the median values of bsfc and specific exhausts emissions are reported in **Table 8**. During almost all cases, the effective work shows low variation in the values obtained during the same operation, while during turns and stops, the unpredictability among the sections is higher, which highlights a lower homogeneity in the characteristics of turns and stops. This is mainly due to the working features.

Table 8. Median values per work state and variable of each of the studied operations. Top: rotary harrowing; Middle: ploughing, sowing and rolling; Bottom: spike harrowing

Variable	**Work state**	**Median**				
		Rotary harrowing				
		Headland strategy A	Headland strategy B	Headland strategy C	Headland strategy D	Headland strategy E
bsfc	Eff. work	308.7	309.8	310.4	309.7	310.5
	Turns	322.9	349.7	348.2	384.4	342.2
	Stops	328.5	--	414.0	446.8	304.5
CO_2	Eff. work	1027.3	1034.3	1034.2	1019.8	1029.0
	Turns	1121.5	1167.7	1177.3	1304.2	1144.9
	Stops	1320.6	--	1324.0	1512.0	1032.3
NO_X	Eff. work	7.9	7.0	7.4	7.7	7.0
	Turns	9.0	9.4	10.2	10.3	9.5
	Stops	10.6	--	9.5	8.7	12.8
CO	Eff. work	0.4	0.3	0.2	0.1	0.2
	Turns	0.6	0.4	0.4	0.3	0.3
	Stops	0.7	--	0.2	0.0	0.5

Variable	**Work state**	**Median**				
		Ploughing		Sowing		Rolling
		Section 1	Section 2	Section 1	Section 2	
bsfc	Eff. work	271.4	273.5	226.7	225.2	220.2
	Turns	303.7	316.5	232.3	231.8	220.5
	Stops	321.1	332.8	251.3	277.9	303.9
CO_2	Eff. work	913.0	908.4	756.1	750.9	733.0
	Turns	1056.8	1053.7	769.2	747.5	738.5
	Stops	1098.3	1100.4	--	943.3	1001.1
NO_X	Eff. work	8.5	7.8	9.2	9.3	7.6
	Turns	12.3	10.3	9.0	8.8	7.5
	Stops	10.8	9.7	--	8.0	8.8
CO	Eff. work	0.0	0.1	0.0	0.1	0.0
	Turns	0.0	0.2	0.0	0.1	0.0
	Stops	0.0	0.0	--	1.6	0.4

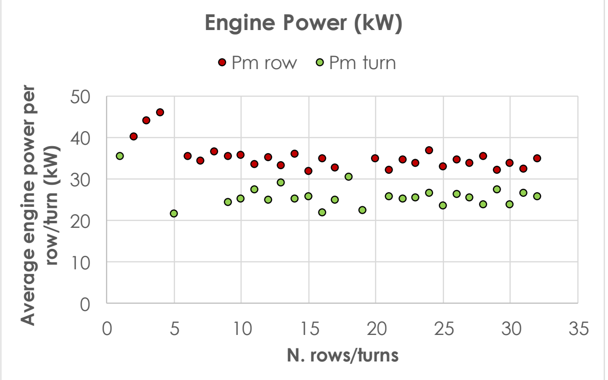
Variable	**Work state**	**Median**			
		Spike harrowing			
		Section 1	Section 2	Section 3	Section 4
bsfc	Eff. work	247.9	211.2	254.0	307.2
	Turns	238.4	213.0	252.5	301.1
	Stops	331.1	277.0	303.1	285.7
CO_2	Eff. work	804.4	699.1	846.0	1028.5
	Turns	785.5	713.8	837.0	1016.5
	Stops	1241.7	862.6	1059.3	944.9
NO_X	Eff. work	5.4	6.3	5.7	5.9
	Turns	5.7	6.0	5.4	5.9
	Stops	9.3	7.0	10.6	7.8
CO	Eff. work	0.4	0.0	0.0	0.1
	Turns	0.3	0.0	0.0	0.1
	Stops	0.1	0.8	0.1	0.1

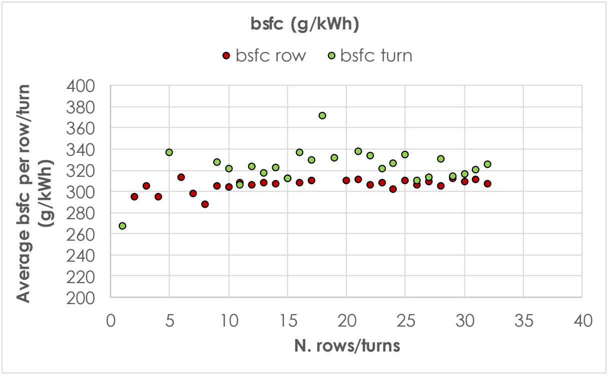
The data processing phase also allowed recognising every single row of effective work and every turn, which permitted to identify an average value for every working state of every operation. **Figure 32** reports these results for engine power (kW), brake specific fuel consumption (g/kWh) and specific gases emissions of CO_2, CO and NO_X (g/kWh) as well as the average value of fuel consumption (dm^3/h) and of exhaust gas emission (g/h).

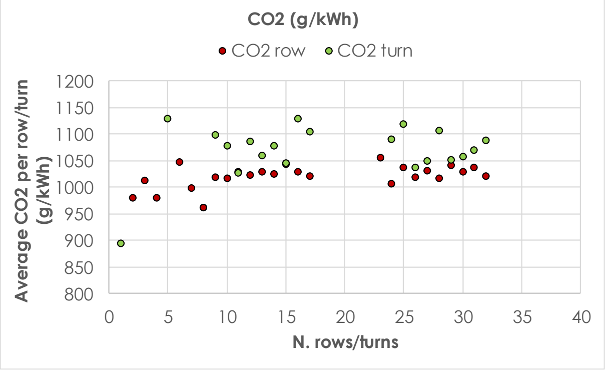
In more details, every dot represents each complete row and each complete turn and the resulting value is the average of that single row and turn. The trend between the way forth and the way back can be explained by specific field-soil conditions as well (such as during ploughing section 2 when the ways forth and back show consistent ups and downs). Although every operation is characterised by specific working assumptions (i.e. engine speed and torque) that make all operations independent from each other, as expected, in almost all cases the developed engine power (kW) is higher during the effective work rather than during the turns. This is shown clearly during rotary harrowing, ploughing section 1 and rolling, while during the other operations the distinction is restrained. Higher engine power during effective work also explains why bsfc and specific emissions (g/kWh) are mostly lower during the effective work than during the turns at the headlands. The lack of some data on emissions is due to the rinsing phases of the instrument. The most unpredictable behaviour is due to CO, where uncontrollable variables have an important role (e.g., O_2 concentration).

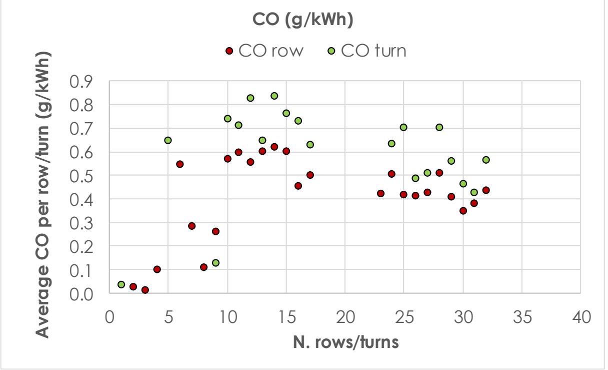
Rotary harrow - Headland A

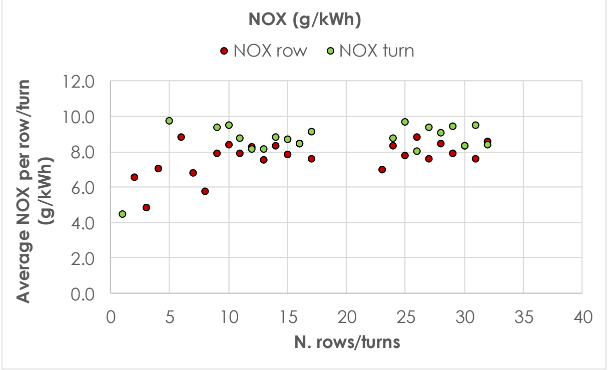
Average values			
Variable		Row	Turn
Fuel	dm³/h	13.5	10.5
CO_2	g/h	36259.3	27567.4
CO	g/h	14.0	14.9
NO_X	g/h	272.7	222.7





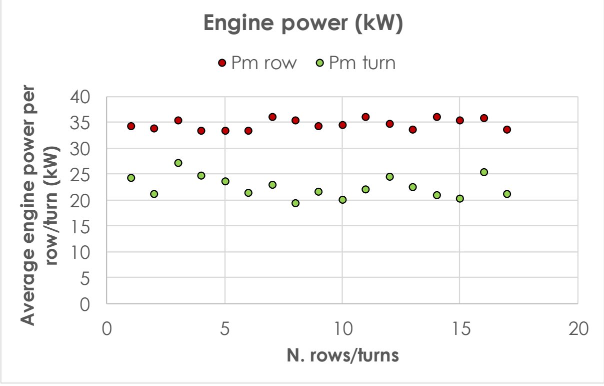


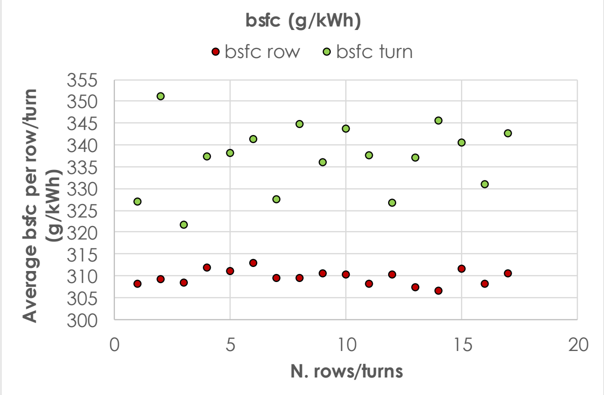


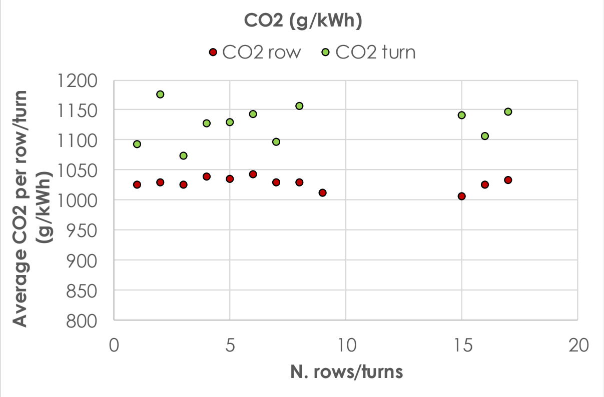


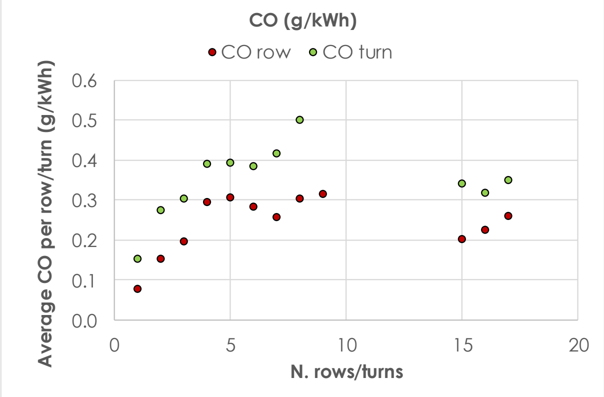
Rotary harrow - Headland B

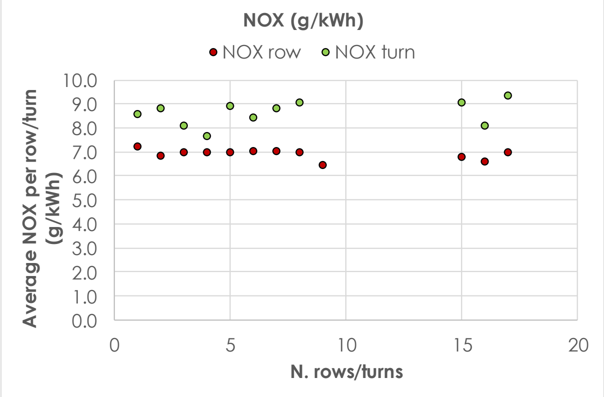
Average values			
Variable		Row	Turn
Fuel	dm^3/h	13.4	9.4
CO_2	g/h	35337.5	25550.1
CO	g/h	8.2	7.8
NO_X	g/h	237.5	195.4





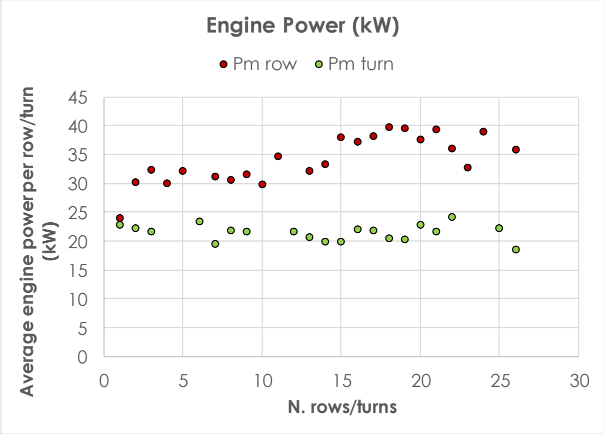


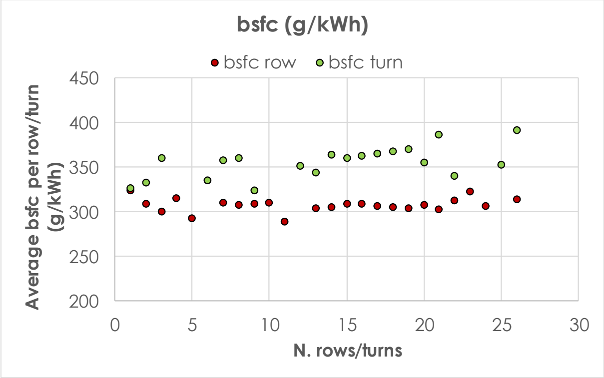


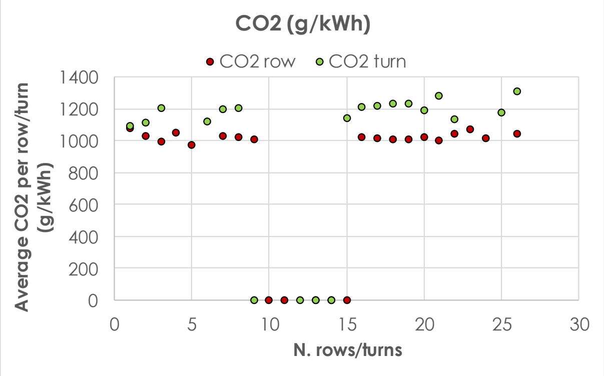


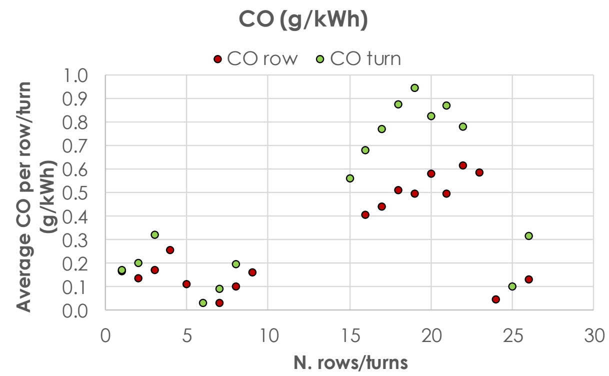
Rotary harrow - Headland C

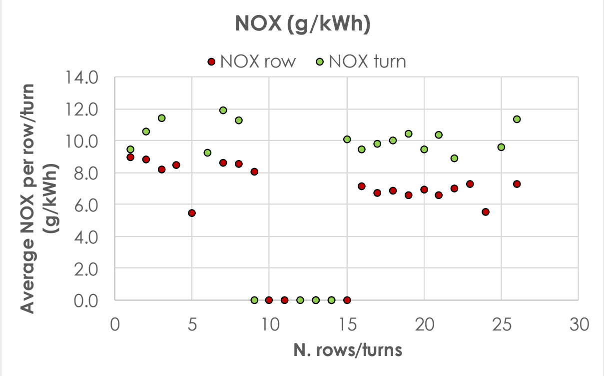
Average values			
Variable		Row	Turn
Fuel	dm³/h	13.1	9.5
CO_2	g/h	35016.0	25559.0
CO	g/h	10.8	10.4
NO_X	g/h	249.5	218.4





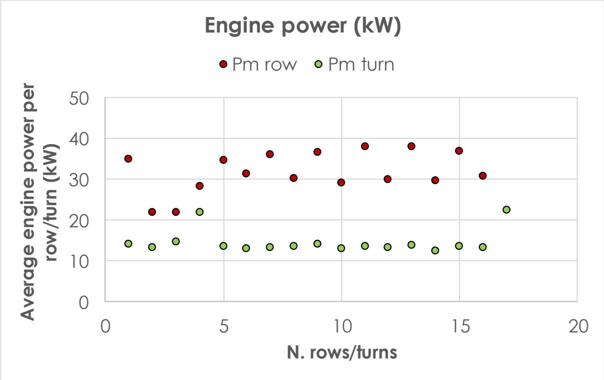


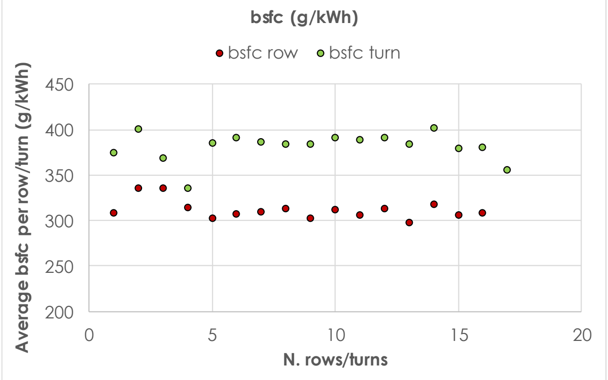


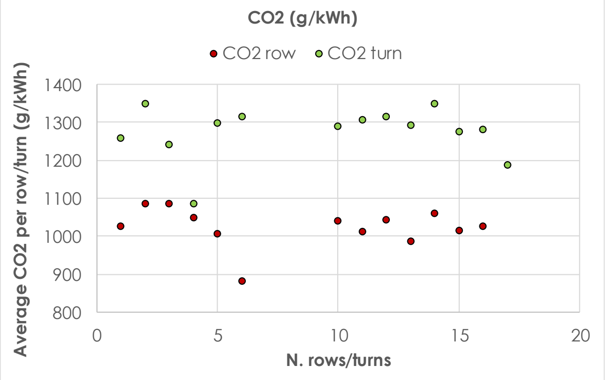


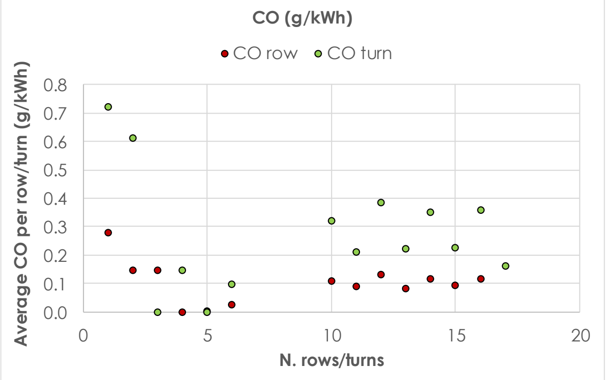
Rotary harrow - Headland D

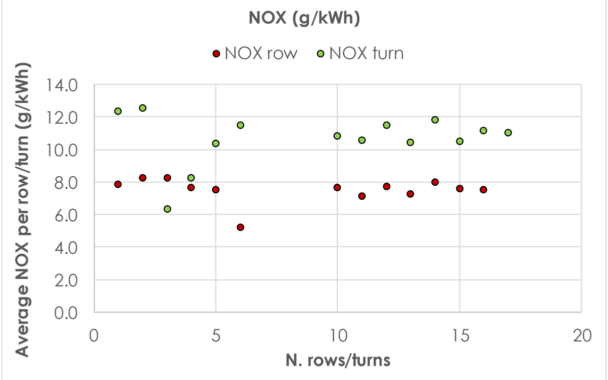
Average values			
Variable		Row	Turn
Fuel	dm^3/h	12.4	6.9
CO_2	g/h	31903.1	18671.8
CO	g/h	3.2	3.9
NO_X	g/h	233.3	155.8





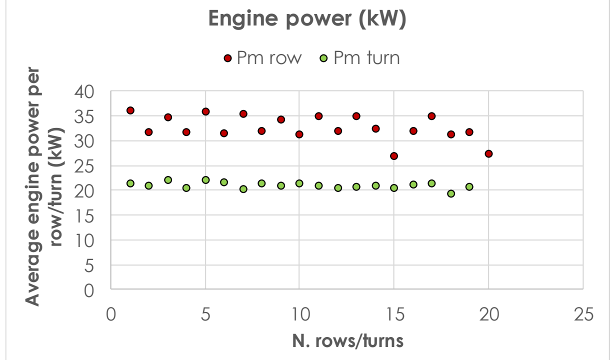


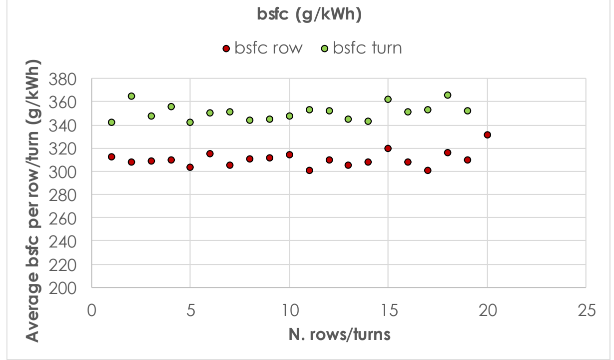


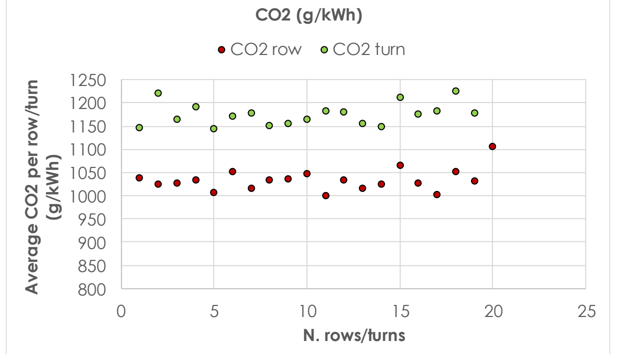


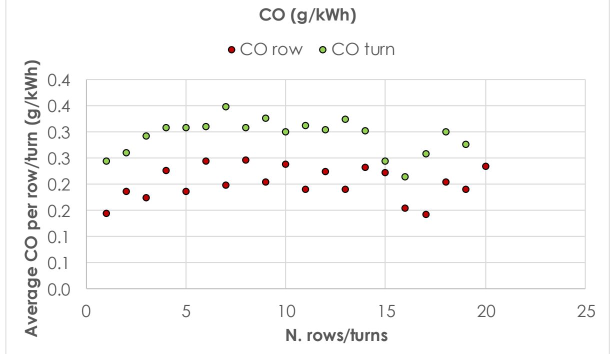
Rotary harrow - Headland E

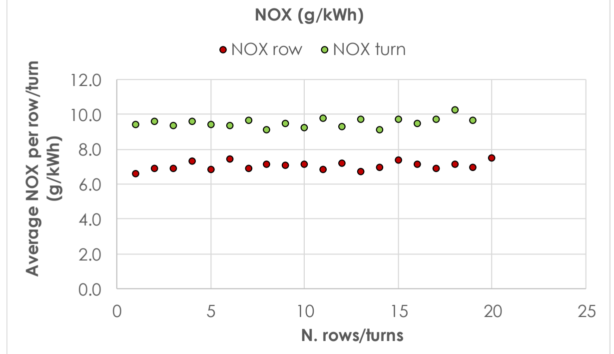
Average values			
Variable		Row	Turn
Fuel	dm^3/h	12.7	9.2
CO_2	g/h	33813.1	24728.2
CO	g/h	6.6	6.1
NO_X	g/h	230.5	200.5





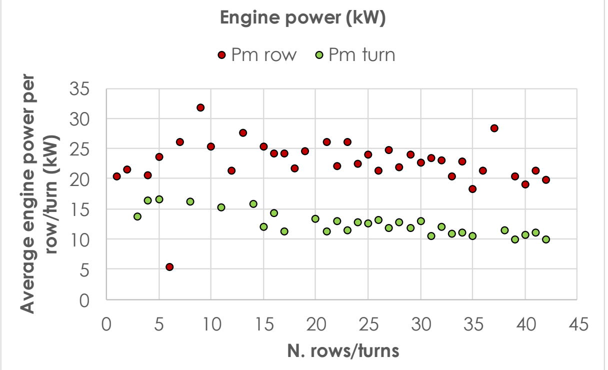


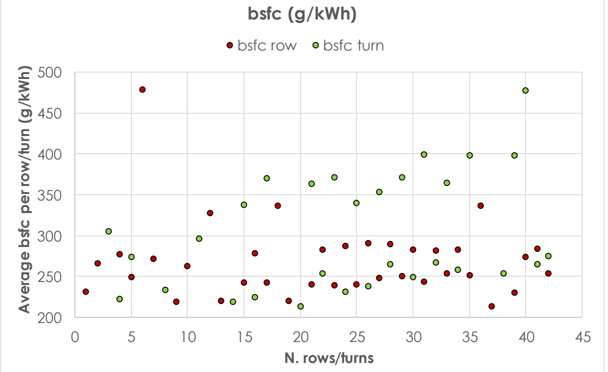


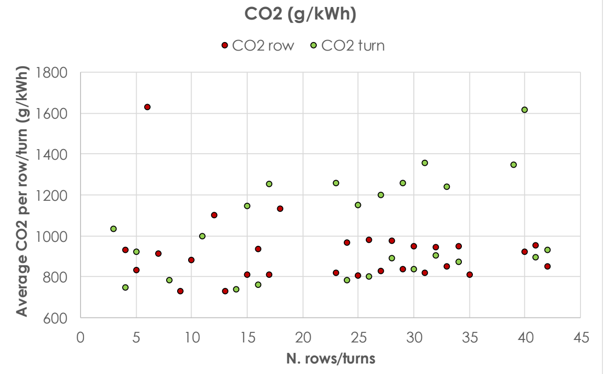


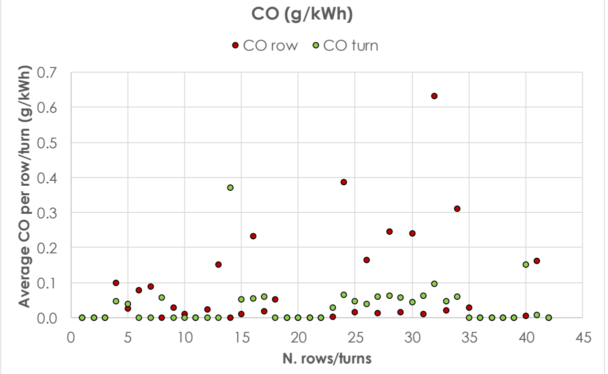
Ploughing 1 (H_1 = 18 cm)

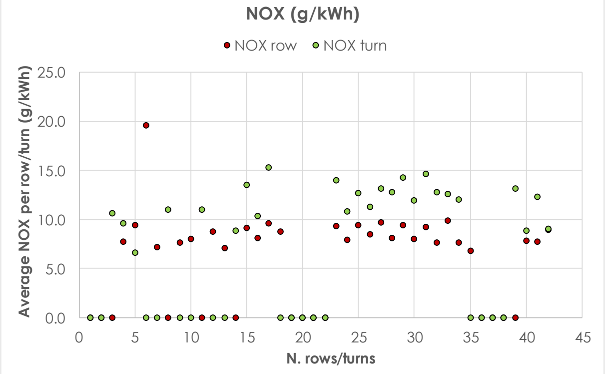
Average values			
Variable		Row	Turn
Fuel	dm³/h	7.5	4.7
CO_2	g/h	20244.2	12836.3
CO	g/h	2.5	0.8
NO_X	g/h	193.5	147.1





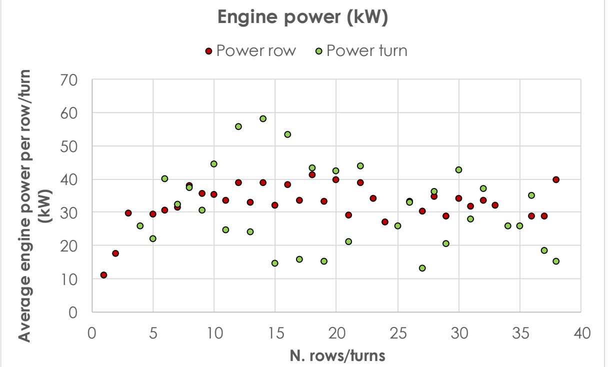


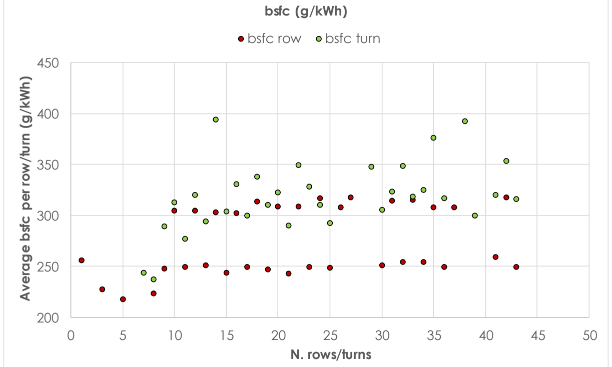


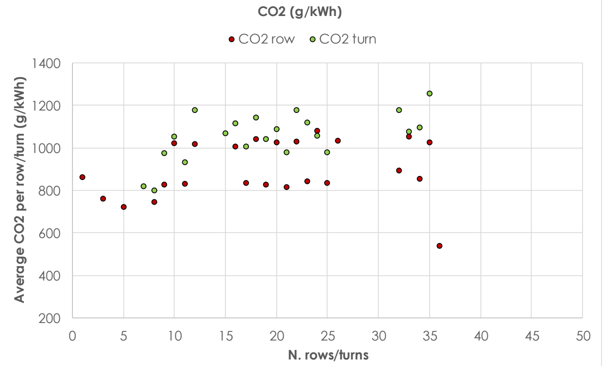


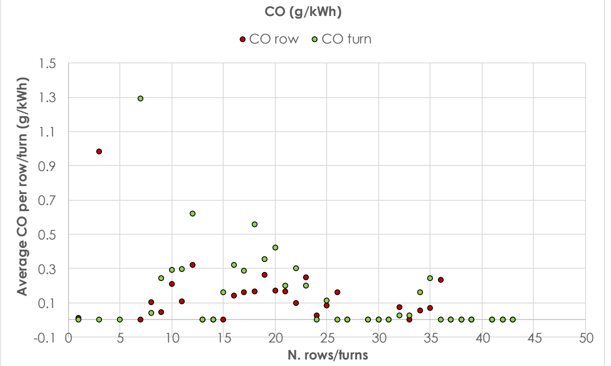
Ploughing 2 (H_2 = 28 cm)

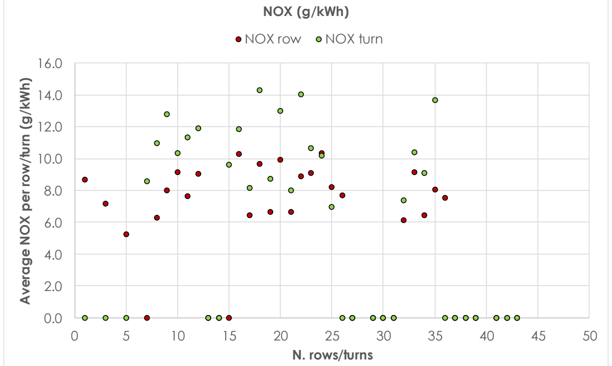
Average values			
Variable		Row	Turn
Fuel	dm³/h	11.2	5.8
CO_2	g/h	29210.9	15888.9
CO	g/h	9.0	4.4
NO_X	g/h	259.9	156.4





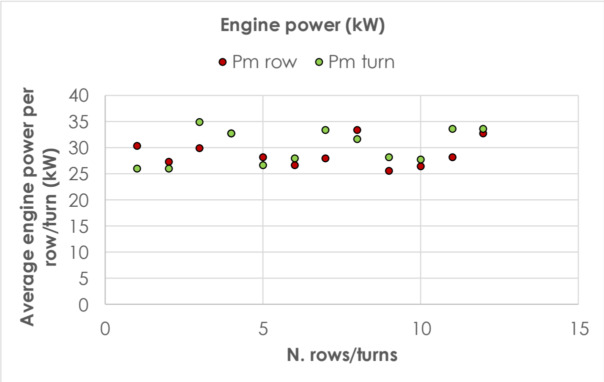


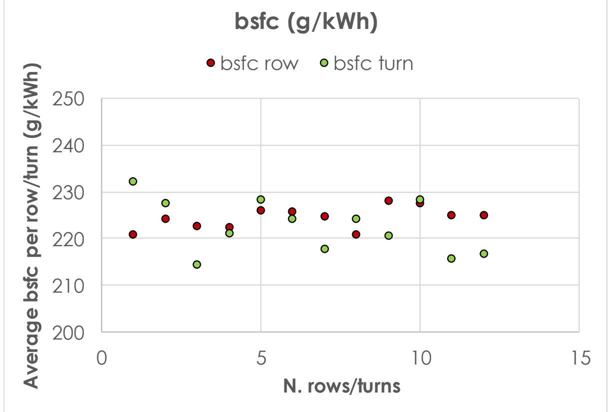


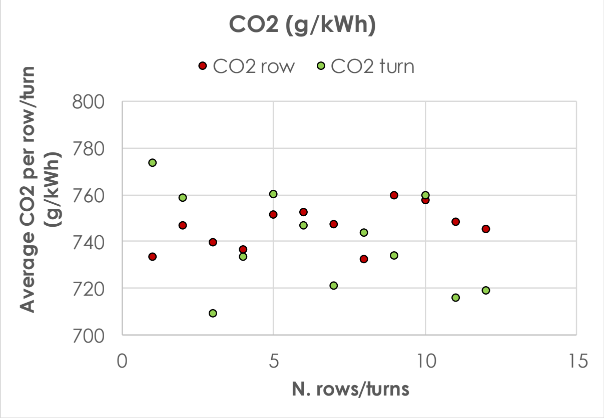


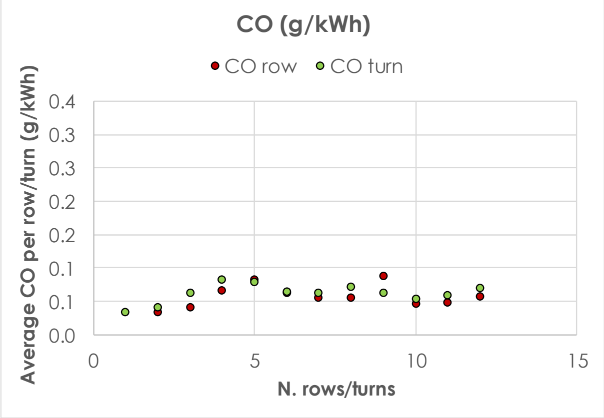
Sowing section 1 (outer)

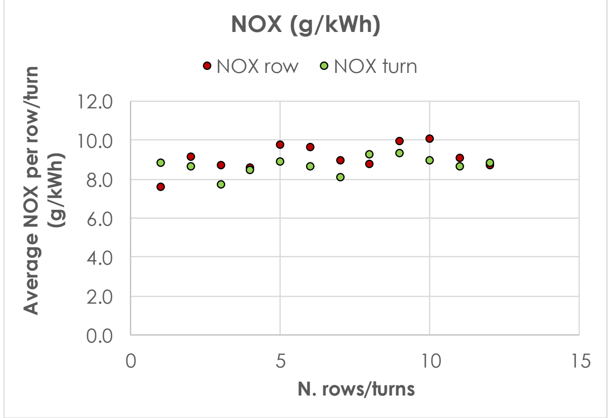
Average values			
Variable		Row	Turn
Fuel	dm^3/h	8.2	8.4
CO_2	g/h	21696.4	22286.9
CO	g/h	3.3	1.9
NO_X	g/h	263.7	262.4





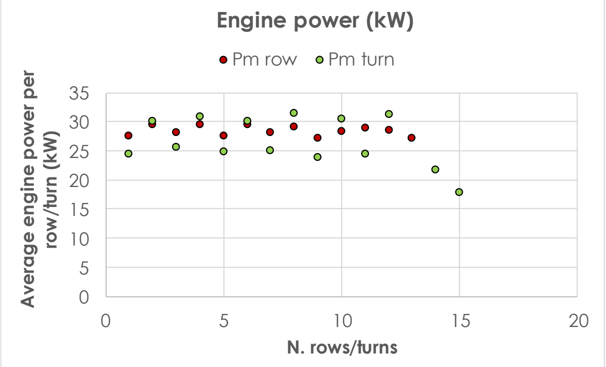


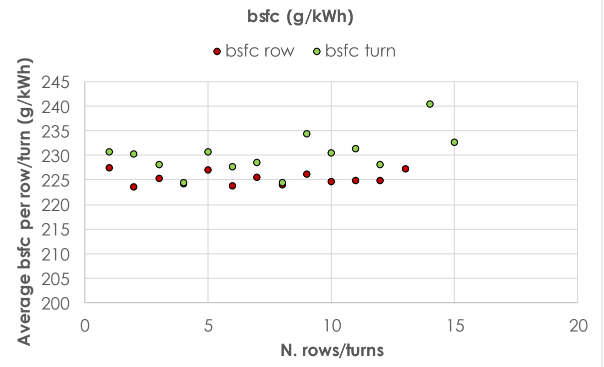


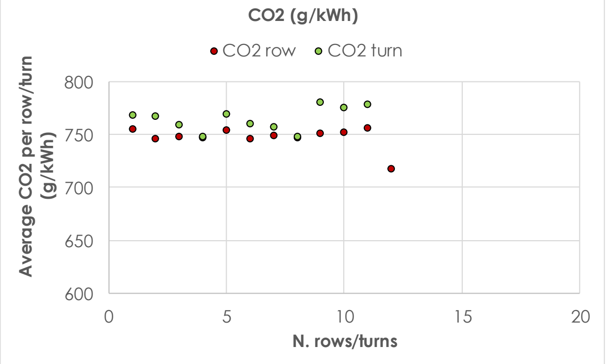


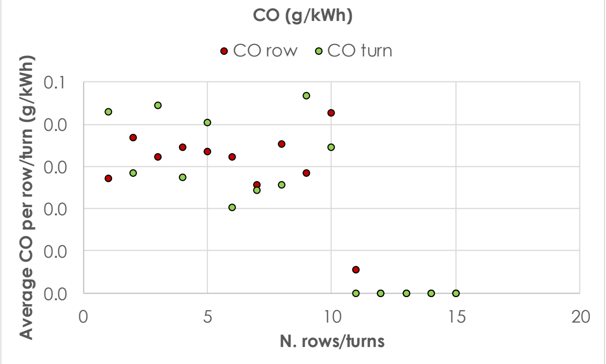
Sowing section 2 (inner)

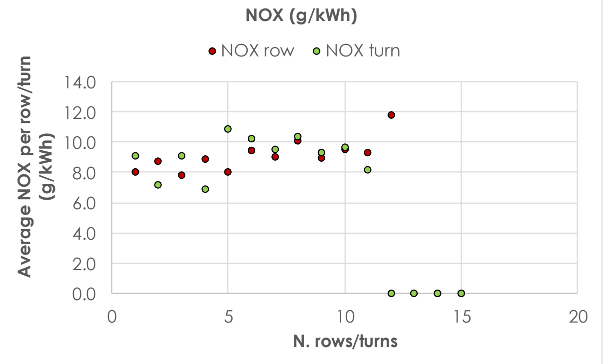
Average values			
Variable		Row	Turn
Fuel	dm^3/h	8.0	7.7
CO_2	g/h	21421.4	21063.2
CO	g/h	0.8	0.8
NO_X	g/h	262.2	251.0





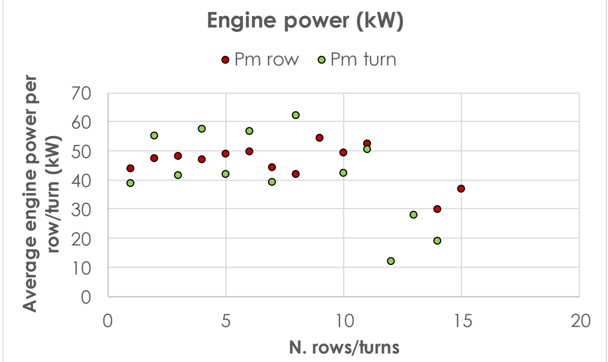


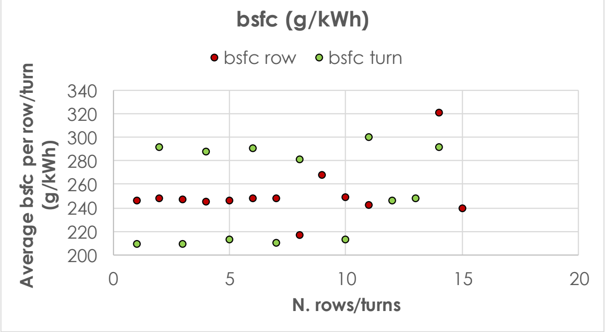


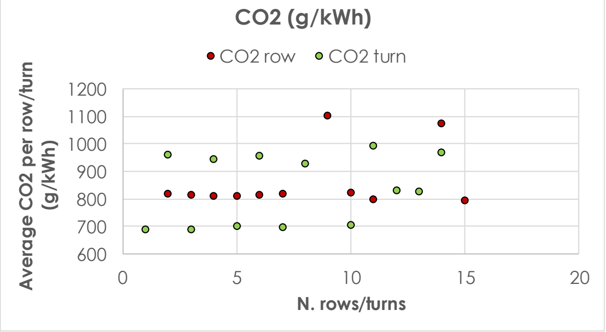


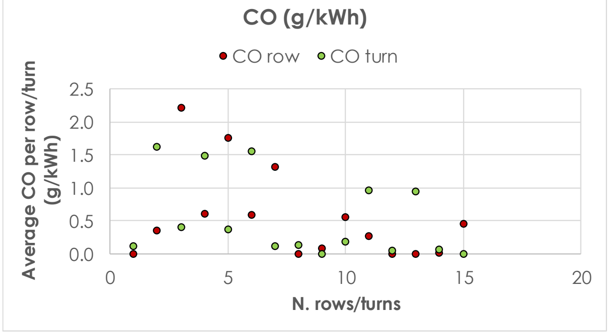
Harrowing 1

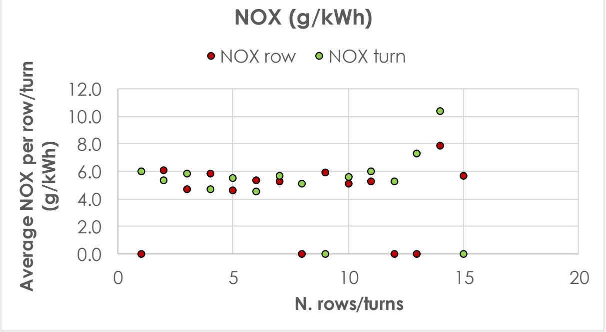
Average values			
Variable		Row	Turn
Fuel	dm³/h	14.4	13.6
CO_2	g/h	39894.0	35769.7
CO	g/h	35.5	30.7
NO_X	g/h	256.6	237.4


Engine power (kW)


bsfc (g/kWh)

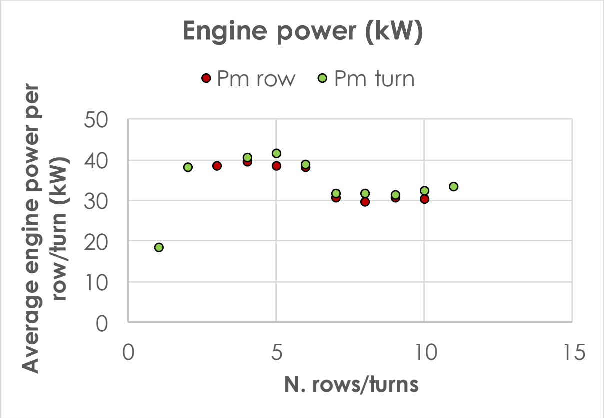

CO2 (g/kWh)

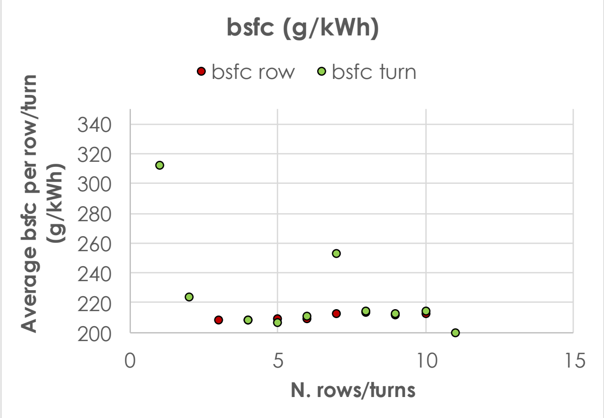

CO (g/kWh)

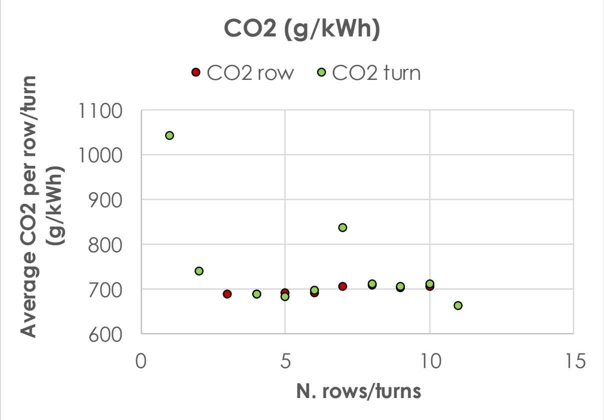

NOX (g/kWh)

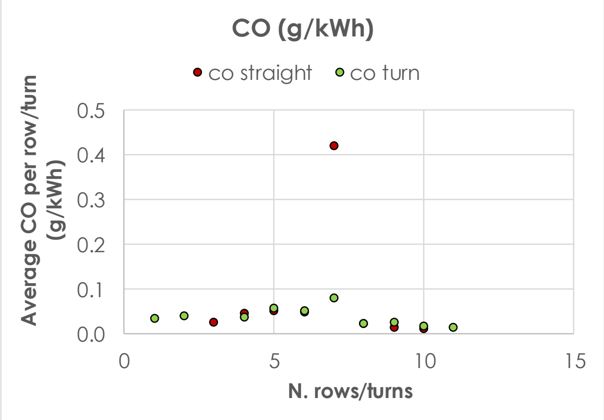
Harrowing 2

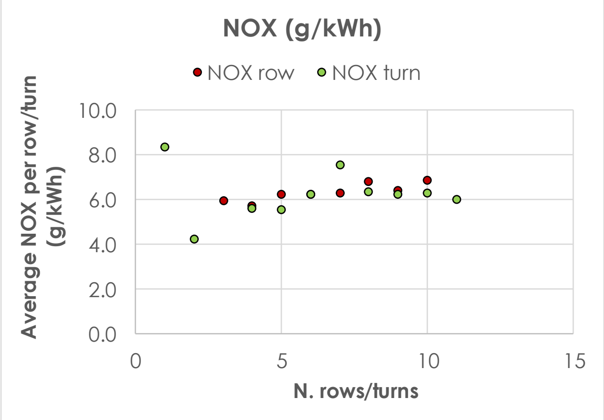
Average values			
Variable		Row	Turn
Fuel	dm³/h	9.1	9.3
CO_2	g/h	23990.7	24716.0
CO	g/h	2.6	1.3
NO_X	g/h	216.1	205.1





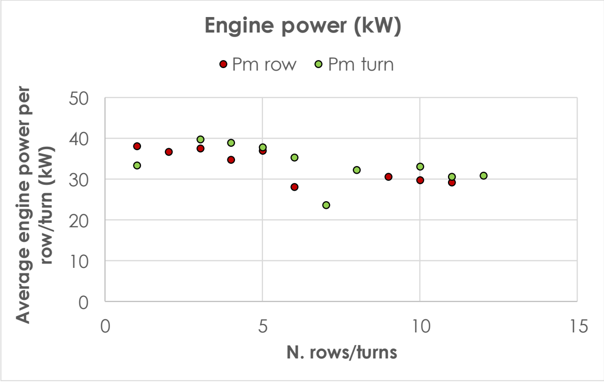


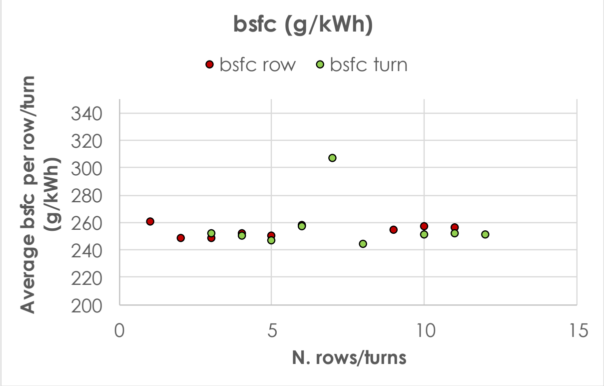


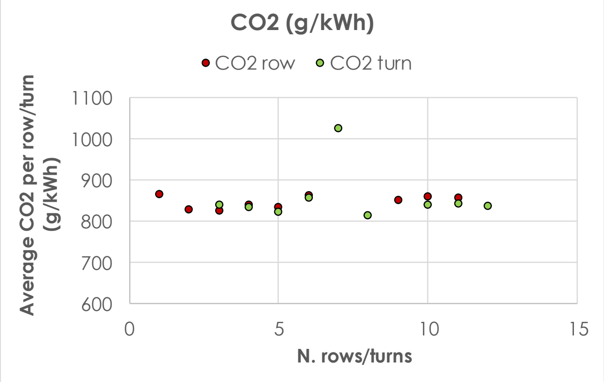


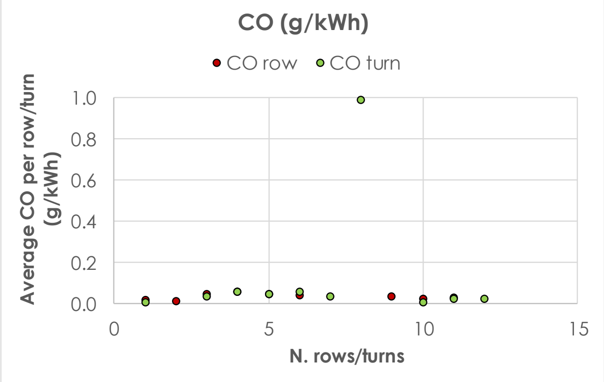
Harrowing 3

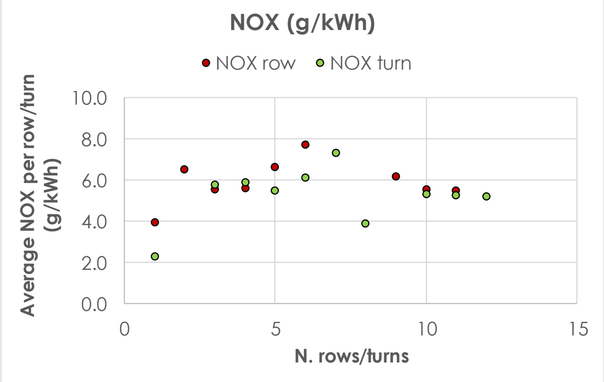
Average values			
Variable		Row	Turn
Fuel	dm^3/h	10.6	10.0
CO_2	g/h	28386.9	26728.4
CO	g/h	1.1	4.2
NO_X	g/h	196.6	175.4





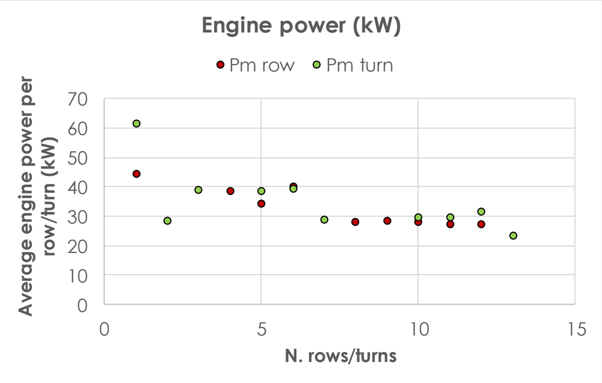


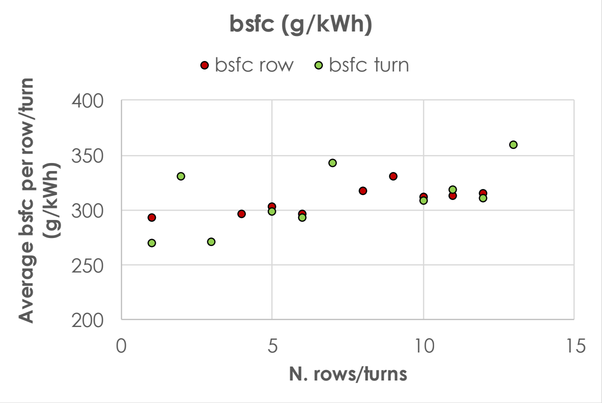


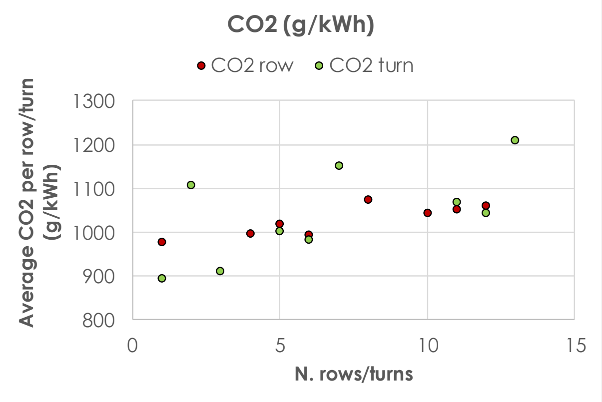


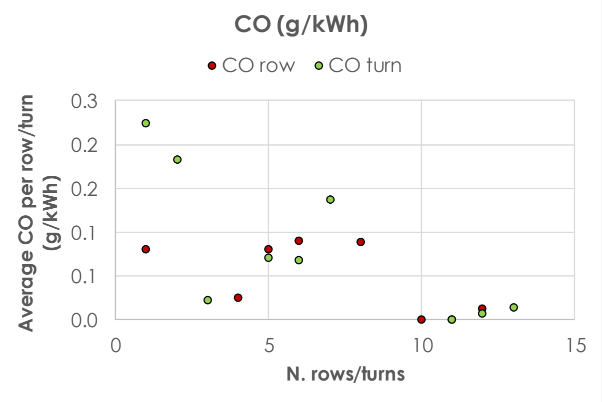
Harrowing 4

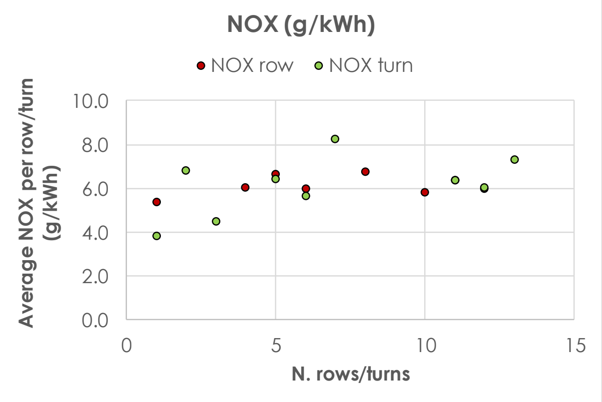
Average values			
Variable		Row	Turn
Fuel	dm³/h	12.6	13.3
CO_2	g/h	34223.0	36142.4
CO	g/h	1.7	3.3
NO_X	g/h	203.1	206.8

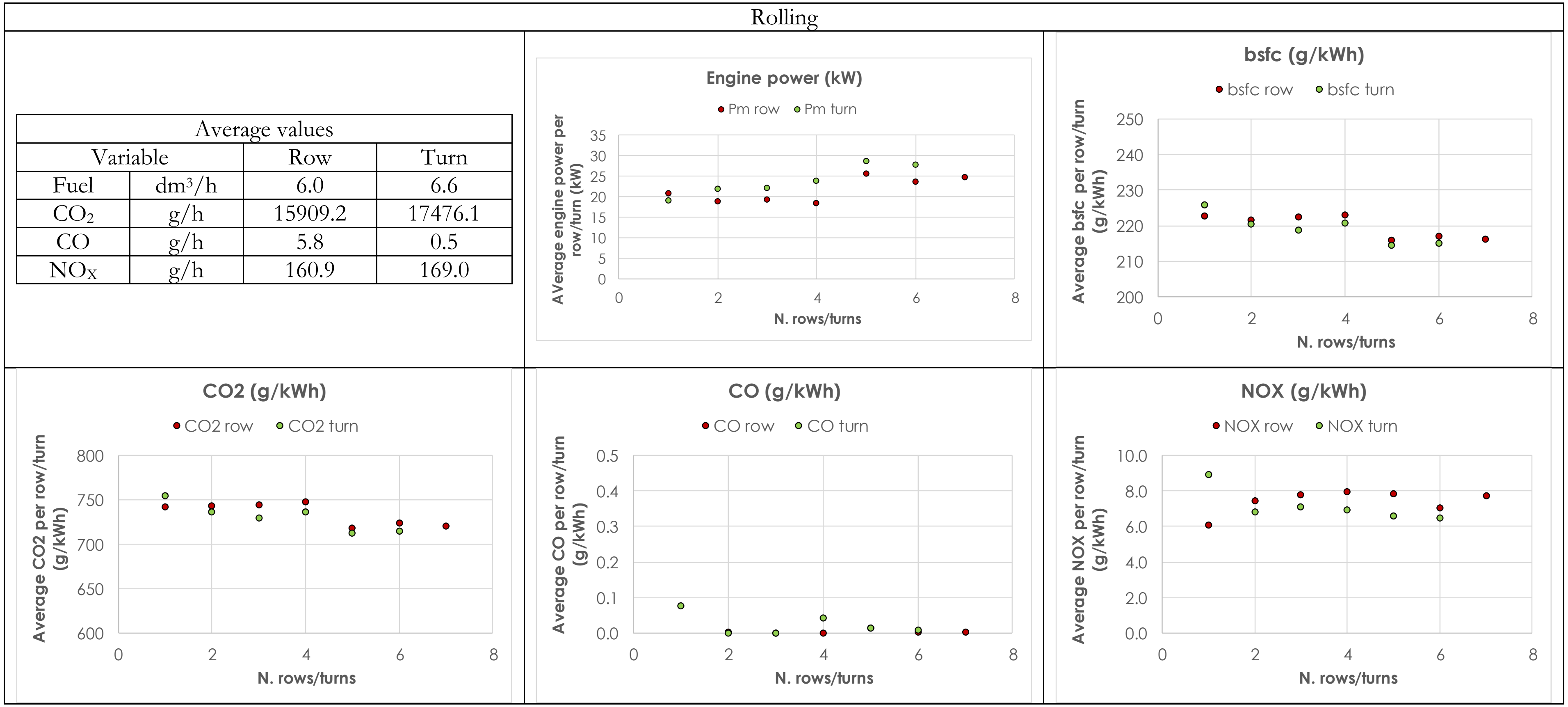











Rolling

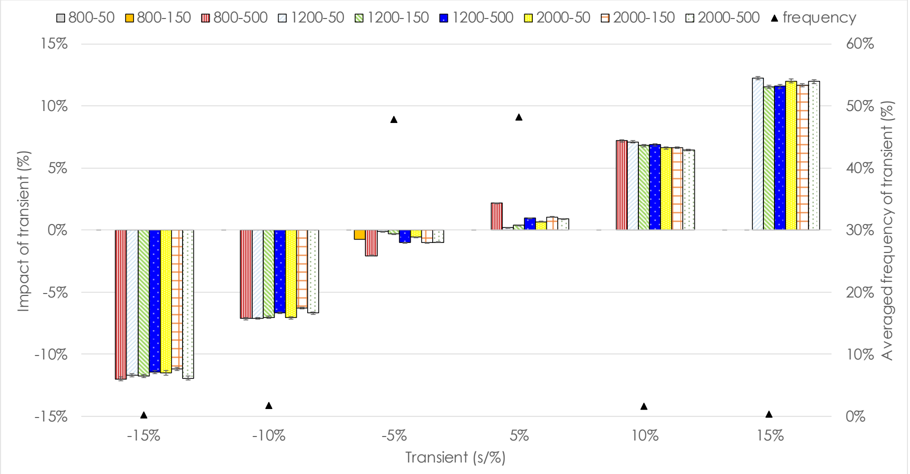
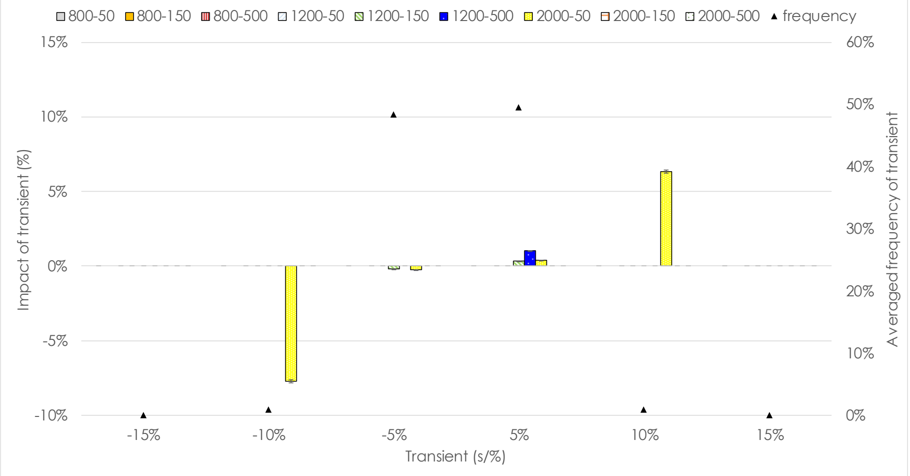
Average values			
Variable		Row	Turn
Fuel	dm^3/h	6.0	6.6
CO_2	g/h	15909.2	17476.1
CO	g/h	5.8	0.5
NO_X	g/h	160.9	169.0

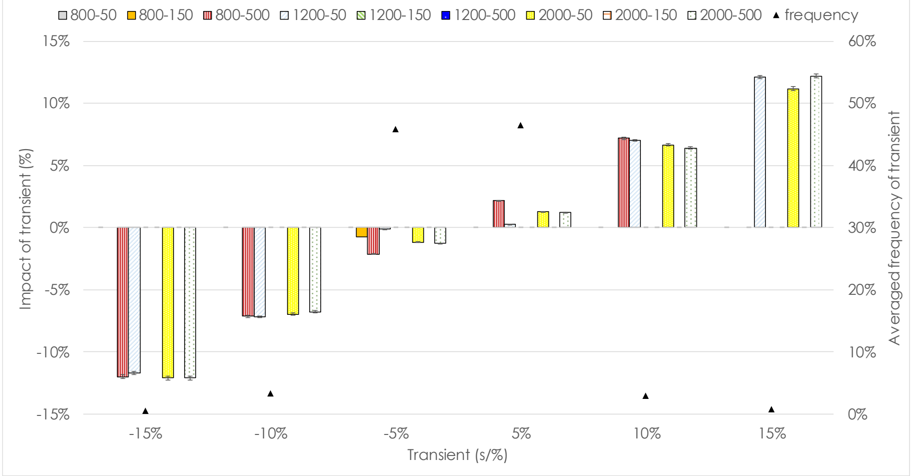
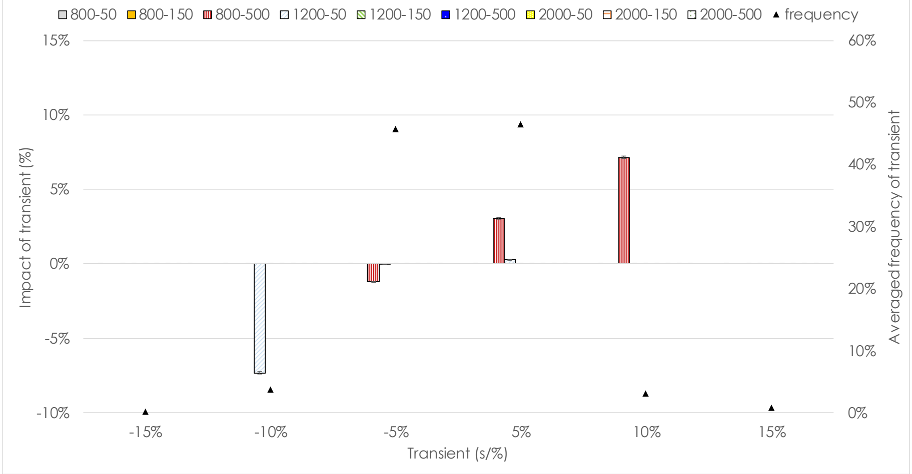
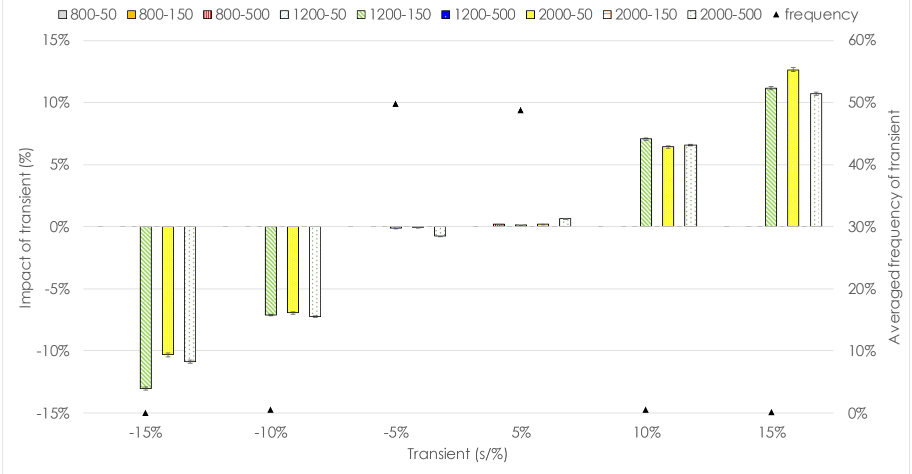
Figure 32. On the top: average absolute values per row of effective work and per turn at headlands of fuel consumption (dm^3/h) and engine exhaust emissions (g/h), developed engine power (kW) and brake specific fuel consumption (bsfc; g/kWh). On the bottom: CO_2 specific emission (g/kWh), CO specific emission (g/kWh) and NO_X specific emission (g/kWh).

In **Figure 33** is reported the impact of the transients during all operations together (A) and during each operation (B). The histograms describe the combination of engine speed and torque as follows:

- engine speed is grouped as s < 800 rpm, 800 < s < 1200 rpm, s > 1200 rpm;
- torque is grouped as M < 50 Nm, 50 < M < 150 Nm, M > 150 Nm.

In sections named "(B)", few histograms are available per operation, since engine speed was defined as starting assumption and, therefore, not every group was available.

All operations (A)	
Rotary harrowing (B)	

Ploughing (B)	
Sowing (B)	
Spike harrowing (B)	

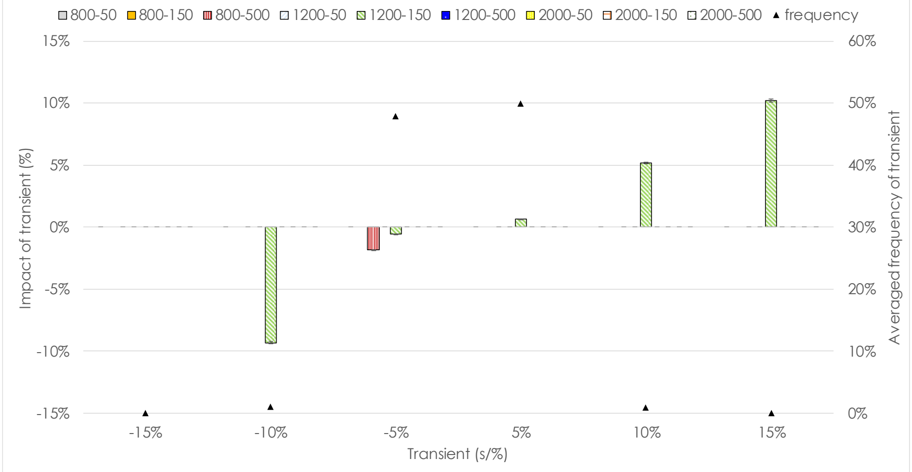
Rolling (B)	

Figure 33. Impact of the transient and frequency of transients on all studied operations together (A) and split per operation (B). The legend reports the combination of values of engine speed (<800 rpm; 800-1200 rpm; >1200 rpm) and torque (<50 Nm; 50-150 Nm; >150 Nm) per series. Triangle-dots show the averaged frequency of transients.

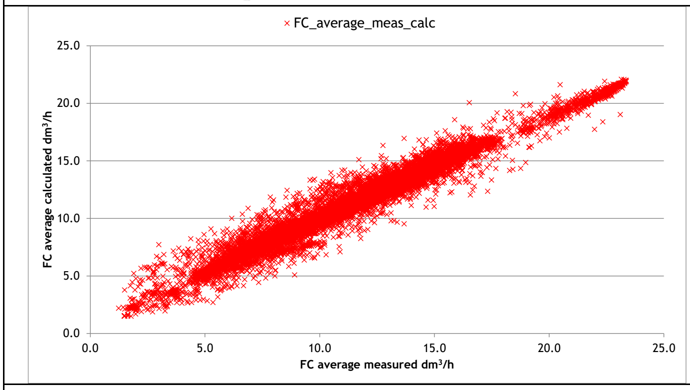
On the X-axis is reported the transient (s/%), which is the rate of change of torque and engine speed per time unit (s) respect to a steady state condition where the transient is zero. On the Y-axis (left) is shown the impact of the transients, showing that restrained transients cause restrained effects on the calculated values (by the model) respect to the measured ones. Instead, when the transient increases in absolute terms, the discrepancy with the steady state calculation grows. On the Y-axis (right) is also reported the frequency of the transient values, from which can be observed that data with higher impact of transients are zero or close to zero; thus, the absolute effect of transients in the studied operations is very low.

From the figure, it can be noticed that most data (96%) are enclosed within -5% and +5% of impact of transient, thus meaning that the effect of transients is restrained along all assessed operations. This impact gets more important with transients around ≤ -10% and ≥ +10%, but the data included in this range are particularly few, as can be gathered from the dots describing data frequency. Ploughing and spike harrowing are the operations showing the highest role of transients, which can be found also in the range ≤ -15% and ≥ +15%. This result confirms what can be expected from ploughing: this operation is one of the most energy consuming, commonly characterised by high engine loads and high possibility of encountering transient conditions (variations in engine speed and torque). For what regards the spike harrowing, the reason is related to how the field trial was performed: in fact, the first section of this operation was characterised by a field divided in three parts on the length, thus involving frequent changes in engine speed and torque on every row. Still, their role is restrained due to the fact that this condition occurred on one of the four studied sections, therefore, only part of the data is affected by this.

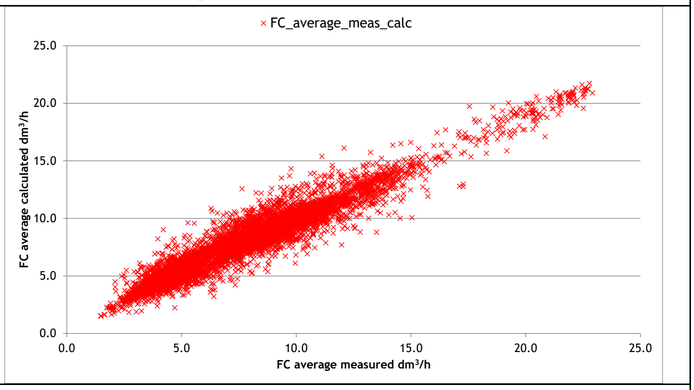
If the transient equation by Lindgren (2005) is introduced, it can be hypothesised to have even better reductions between recorded and measured values respect to Lindgren (2005) study, where the presence of transient factors was helpful to reduce the recorded-calculated differences from 30% to approximately 5%.

Concerning the prediction model used for the engine of Valtra N101, its response was very good, especially for fuel and CO_2. In **Figure 34** are reported the curves and the coefficients of determination R^2 for all variables and distinguished in effective work, turns at headlands and stops.

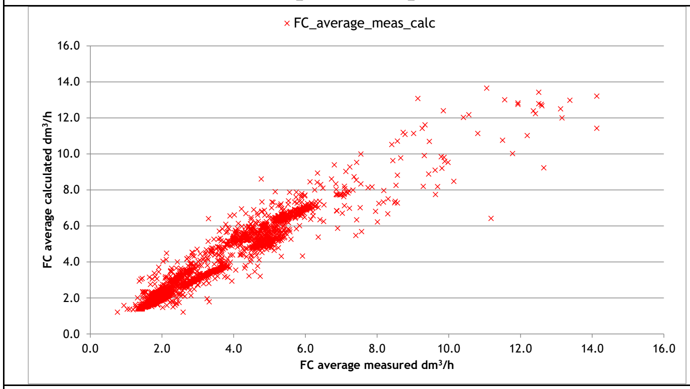
Fuel consumption; Effective work ($R^2 = 0.97$)



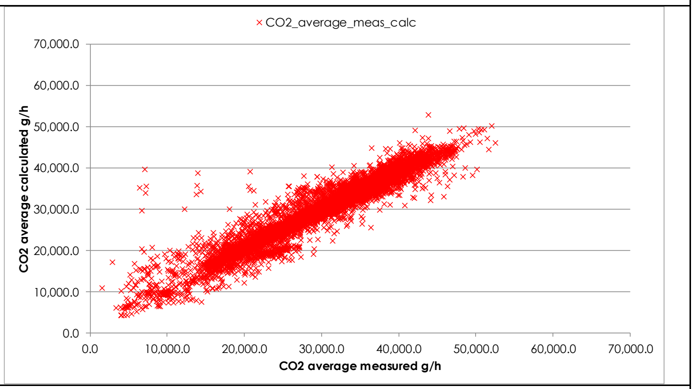
Fuel consumption; Turns at headlands ($R^2 = 0.92$)



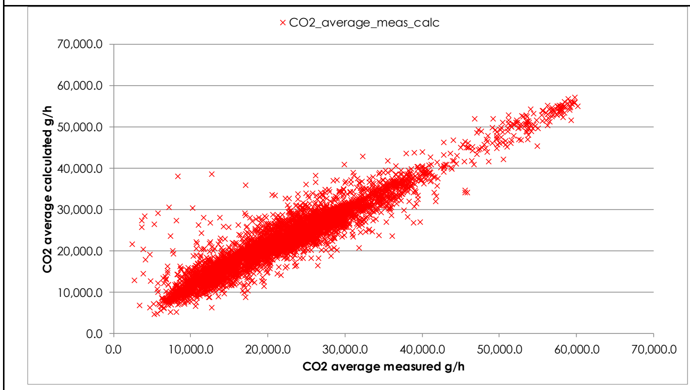
Fuel consumption; Stops ($R^2 = 0.95$)



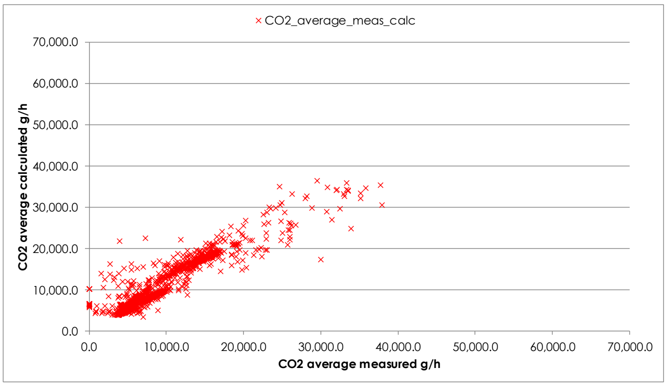
CO_2 emission; Effective work ($R^2 = 0.90$)



CO_2 emission; Turns at headlands ($R^2 = 0.77$)



CO_2 emission; Stops ($R^2 = 0.65$)



NO_X emission; Effective work ($R^2 = 0.22$)

NO_X emission; Turns at headlands ($R^2 = 0.38$)

NO_X emission; Stops ($R^2 = 0.42$)

CO emission; Effective work ($R^2 = 0.19$)

CO emission; Turns at headlands ($R^2 = 0.32$)

CO emission; Stops ($R^2 = 0.05$)

Figure 34. Example of model response for fuel consumption, CO_2, NO_X and CO assessment.

The measurements carried out during this study permitted to identify and understand the real work of the tractor engine Valtra N101 in the identified field conditions. There are several interesting points and progresses obtainable with this typology of data collection:

- on the environmental point of view, the availability of trustworthy data permits to have reliable data for environmental impact studies completed with LCA (Lovarelli and Bacenetti, 2017), as well as to have reliable data that describe the engine features and that permit to draw conclusions on the best driving practices, and on the best choices and technologies for reducing fuel consumption and engine exhaust gases emissions;

- on the mechanical point of view, the possibility of studying the effective behaviour on field of any tractor with the described instrumentation shows the engine's (and the tractor's in general) response to the real working conditions. This permits, once more, to identify the best driving practices and the most preferable working conditions, as well as to obtain information on the effective fuel consumption and exhaust gases emissions (e.g., Lindvall et al., 2015). This last point is very important, especially in view of the stringent directives on the exhaust gas emissions limits that currently regulate the steady state measures.

A positive result is also related to the fact that, once data are collected for the tractor on different working conditions, a robust prevision model can be created for predicting the behaviour of the tractor in other non-measured conditions, with other implements and during other operations simply measuring with a dynamometer the needed information. This means that the monitored tractor is mapped in real working conditions and its behaviour can be still reconstructed for other studies.

Similarly to what has been realised by Januleviĉius et al. (2016, 2017), also the not-to-exceed zones can be investigated, in order to identify the best conditions in which to work. This can also be helpful, together with current and under development technology, in introducing new methods and procedures for precision agriculture (Suprem et al., 2013) as well as in helping farmers and contractors to understand at best the potentialities of their machinery in order to exploit effectively the benefits of such studies on the environmental sustainability of agricultural systems.

6.3 WF assessment *(Paper 6-7)*

From the literature review (*Paper 6*) emerged some points:

- often, at least one of the three components (green, blue and grey water) was not assessed. In most cases, the lacking component was WFgrey (**Figure 35**);
- the green component (WFgreen) was almost in all cases assessed following the FAO Papers n. 56 (Allen, 1998); instead, for WFblue and, mostly, for WFgrey not always the application of a consistent method was found;
- the critics on WFblue focused on the lack of information about the irrigation method, the effective irrigation gross volume, and the water stress and scarcity;
- the critics on WFgrey focused on the lack of a clear methodology and on the unclear and unsure information given.

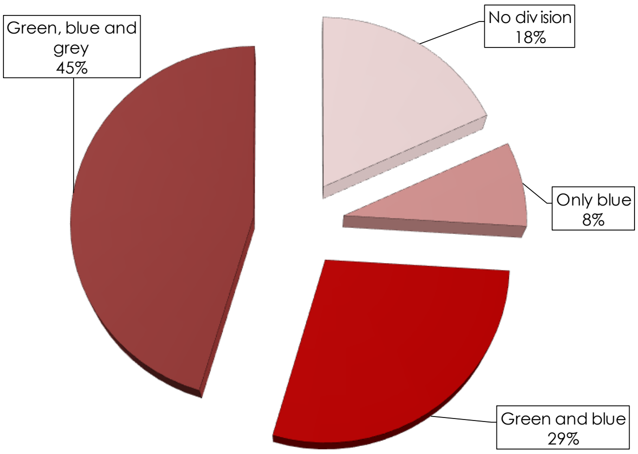
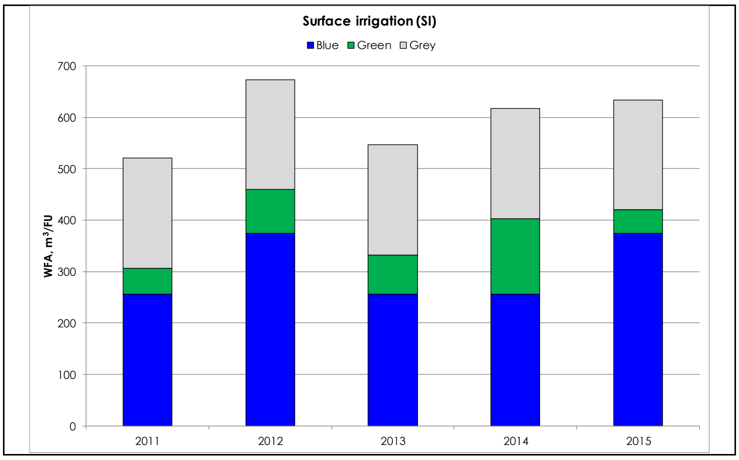


Figure 35. Composition of the WF components available in the reviewed studies.

In order to enhance the WF assessment method, with specific focus on local variability and local studies and on a more detailed assessment for WFblue and WFgrey, the methodological changes proposed during this PhD Thesis brought to realise a case study on a 5-years irrigated maize cultivated in Po Valley (*Paper 7*), where:

- the water balance was quantified (green and blue water) taking into account different irrigation technologies (i.e. surface, sprinkler and drip irrigation);
- the Pollution Water Indicator (PWI) was introduced for studying water pollution.

As expected and as illustrated in **Figure 36**, the most efficient irrigation techniques cause that a lower gross water volume is introduced in the system, thus being closer to the net water demand of the crop as well as to the concept of WFblue (within WFN method). Of course, in cropping seasons with high rainfall, WFgreen is higher and WFblue can be lower, as can be gathered in the figure below for surface and drip irrigation systems.



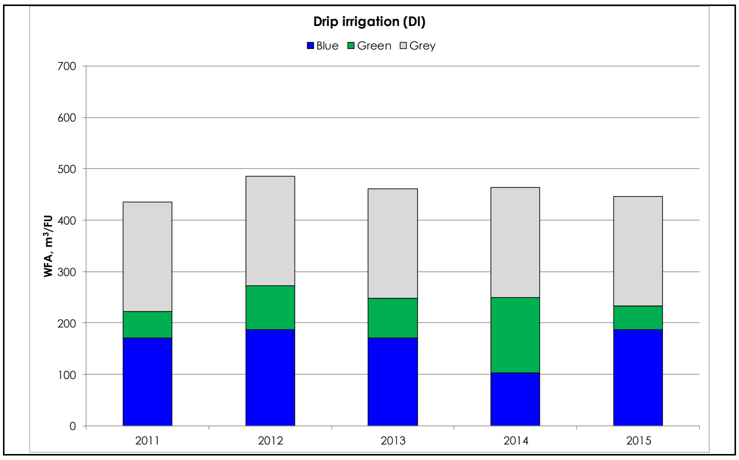


Figure 36. Water Footprint of grain maize in Po Valley 2011-2015 assessed with the WFA method for blue water applied to surface and drip irrigation.

With regard to WFgrey, the WFN method does not give effective information of the environmental perspective. Thus, the PWI is a helpful indicator for introducing the concept of WF together with the LCA one in a wide perspective that regards freshwater environmental sustainability. **Table 9** reports the specific scenarios in which the PWI was studied. The results of its use are shown in **Figure 37** and they show that the most beneficial PWI are related to the cases in which maize straw is collected (i.e. no nutrients leaching on bare soils) (S3) and in which digestate is applied (higher fertiliser efficiency respect to slurry) (S4).

On the opposite, the worst PWI values are obtained with the fast incorporation and direct injection because these solutions involve less ammonia release to air, but higher availability of nutrients in the soil that, in the studied conditions, leach. With the only use of WF indicator (WFgrey), such information could not be retrieved and, instead, would give different outcomes on the WFgrey concerns. The results of this study support those from *Paper 4*.

Table 9. Scenarios in which the PWI was quantified.

Option	Description	Timing of soil incorporation for organic fertilisers	Adopted fertilisers
BS	Incorporation	> 3 days after spreading	Pig slurry and urea
S1	Fast soil incorporation	< 2 h after spreading	
S2	Direct soil injection	During spreading	
S3	Incorporation with straw collection	> 3 days after spreading	Pig slurry and urea + maize straw collection
S4	Incorporation with digestate spreading	> 3 days after spreading	Digestate and urea
S5	Mineral fertiliser - Urea for N	Not applicable	Urea and triple superphosphate

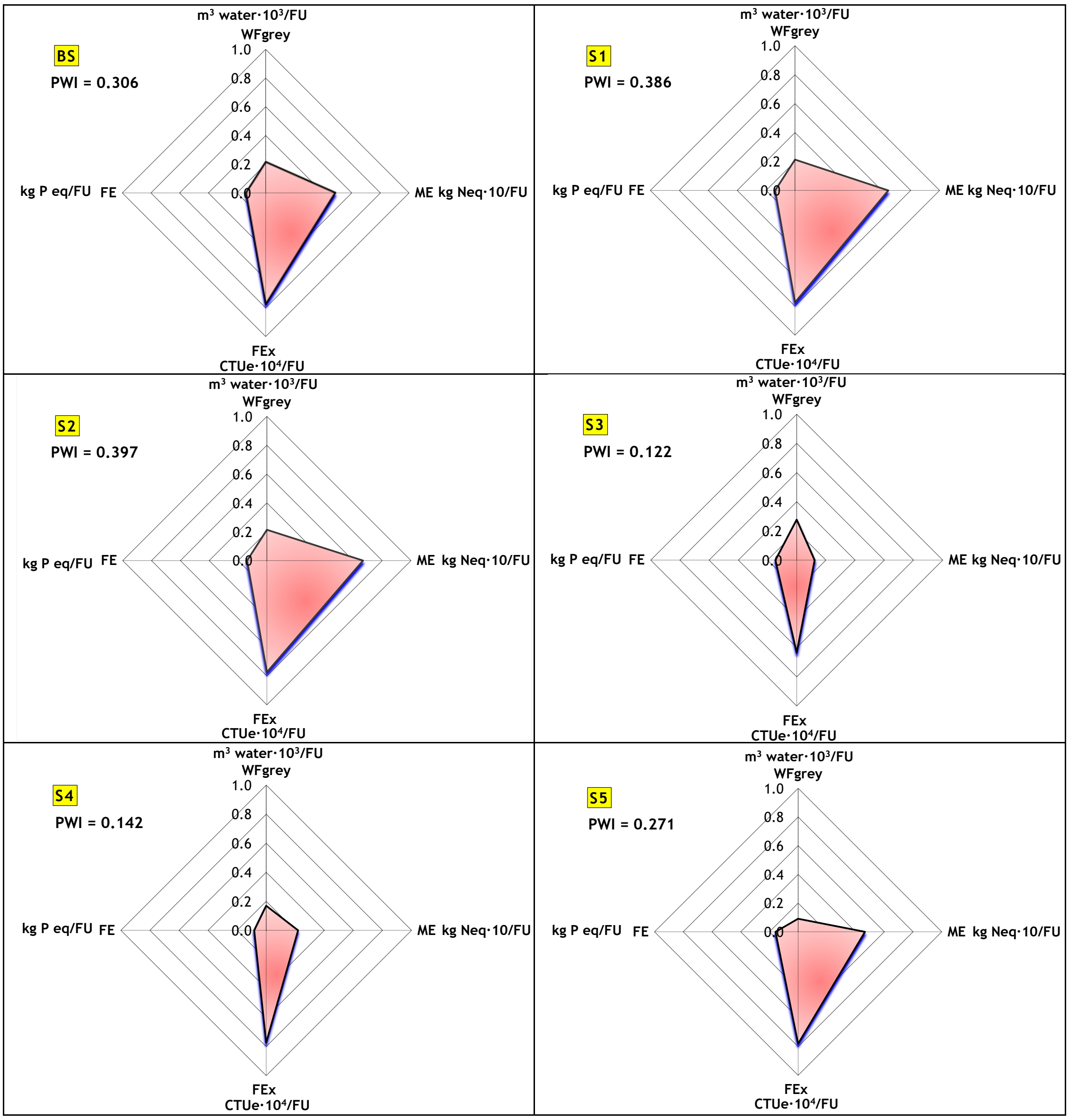


Figure 37. Pollution Water Indicator (PWI) for the analysed scenarios.

The development of PWI was of help in the relation between WF and LCA assessments, given the possibility of focusing on the freshwater resource with a holistic view considering both consumption and pollution.

Currently, widely adopted and accepted methods for WF assessment have been developed within the LCA methodology, with a specific environmental impact category named "Water Depletion". However, the indication resulting from PWI is different from the methods available in LCA software that do not consider the WF with a WFN-view.

7. Conclusions and future research

In the context of environmental systems analyses, several methods are available for environmental assessments, such as the environmental impact assessment and the risk assessment, which have different goals from, for example, the life cycle assessment's ones (Finnveden and Moberg, 2005). In particular, some environmental analysis methods can have a wide applicability (e.g., models) or a local applicability (e.g., environmental impact assessment), but likewise any other scientific scheme play a useful role only when adequately adopted.

LCA is a method to estimate potential environmental impacts in a generalised framework, both widely and locally applicable, that takes into account in the meantime several environmental issues (e.g., global warming, eutrophication, acidification, resources depletion). Expertise is making LCA studies more and more detailed and trustworthy, which is especially important when focusing on productive systems such as the agricultural one. Agriculture, in fact, is characterised by an intrinsic dependency on natural processes and on the variability in productive contexts and local conditions, which causes difficulties in having reliable inventory data. Therefore, specific attention on the inventory data collection is required. Because of this intrinsic variability and interdisciplinarity, the introduction of widely applicable policies can be very difficult for agricultural policy makers, and the use of LCA is important to allow sensible decisions.

In particular, regarding the agricultural machinery field operations, the need of sensible decisions is undeniable to allow introducing policies and/or incentives for developing environmentally sustainable productive systems, for having modern and efficient machinery and reducing the emission of pollutants. In this context, LCA represents the suitable tool to identify and quantify environmental effects, production hotspots and mitigation strategies to undertake. Central achievements regard the possibility of collecting local data and understanding that with LCA is possible to identify and adopt the most beneficial solutions on the environmental perspective, also when several operative alternatives are present. Electronic technology is important in this context, since it simplifies the control of some variables and avoids simplifications and average considerations.

The most significant accomplishment is that the quantification of the environmental load of agricultural machinery operations allows becoming conscious about the role of these production processes often disregarded and developing influential mitigation strategies with a holistic perspective. These strategies focus on broad environmental issues, such as fuel consumption, materials production and wear, emissions of engine exhaust gases, nutrients and active ingredients to air, water and soil, etc. always taking into account not only their use, but also the production and disposal.

In this PhD Thesis, the alternatives displayed to farmers regarding agricultural machinery were studied in order to identify and quantify the differences in environmental impacts due to machinery choices. Having seen that these differences are considerable, further attention to detailed data collection in local contexts resulted an important step forward. The application of modern electronics had a fundamental role and its effect was studied on the data collection during a case study. The results were particularly useful under the two main investigated study areas: (i) LCA and, in particular, LCI completion of agricultural field operations and (ii) mechanisation and alternative machinery selection. Of course, the improvement of inventory reliability must be critically balanced with the increase in time and costs for the primary data collection; however, the benefits arising from such trials are undeniable and are also made easier by the common presence of electronics on recent agricultural tractors (CAN-bus and ISO-bus).

A first trial to make proper inventories was made by developing the tool ENVIAM, which calculates data for the inventory of field operations by considering mechanical variables and local information. ENVIAM could still be improved and kept up-to-date with the continuous uploading of machinery introduced on the market. Moreover, additional improvements could be made with enlarging the groups of machinery included and their coupling features with tractors, as well as with making changes in the input data processing files. For example, with the CAN-bus and exhaust gases analyser data, high specificity in the inventory completion can be reached for every measured operation. These data could be introduced satisfactorily in ENVIAM for additional evaluations on coupling variables. In addition, GPS and GIS tools help with the mapping of fields and with the localisation of the mechanical features on the fields. Nevertheless, it must be kept in mind that the goal is the environmental sustainability assessment and therefore, that the goal is not a too much site-specific analysis, but an environmental sustainability evaluation of the field operation and of the improvements to mitigate the related potential environmental impact. Therefore, tools must be introduced in accordance with the defined goals and scopes. Additionally, it must be kept in mind that LCA involves modelling the potential environmental impacts, and modelling involves uncertainties (Thomassen et al., 2008) that can be reduced but not fully avoided.

The results of this project can be very interesting for the near future of agricultural mechanisation and for making agriculture more and more environmentally sustainable for present and future generations. Manufacturers could introduce similar evaluations in their activities and furnish information to farmers and contractors who are the final users of machinery. With this information, farmers could become aware of their role on the environmental loads of their production systems and understand their potentialities in improving the environmental sustainability of agricultural productions. This can be achieved, for example, by adopting more adequate driving practices and modern technology and paying attention to the engine variables occurring during the different working states, in order to respond at best to the environmental needs of society.

Analysing the sustainability concerns, there is a very recently growing interest in the Life Cycle Sustainability Assessment (LCSA) framework. LCSA is a near-future important application of sustainability evaluations with a life cycle perspective that permits to evaluate not only the environmental aspects as due to the Life Cycle Assessment (LCA), but also the economic (Life Cycle Costing; LCC) and social (Social Life Cycle Assessment; SLCA) ones (Notarnicola et al., 2017).

Another facet related with agriculture is the share of freshwater used and depleted during the production processes, especially at Italian latitudes (and mainly Po Valley ones). In this context, understanding the importance of freshwater resource, of its current use and future limitations and introducing the idea that new technologies are necessary not only for the crop cultivation and for the farmer production costs, but also for the environment, is a very important goal for the scientific community. To this point, both WF and the Pollution Water Indicator (PWI) developed in this Thesis are helpful in informative capabilities for stakeholders, farmers and policy makers. The possibility of taking advantage of LCA holistic view and of WF embedded-water view permits to develop, also for water concerns, mitigation strategies and environmentally sustainable production conditions.

The possibility of effectively introducing WF in the LCA framework is valuable for the ease of both data collection and calculation of full sustainability assessments. However, it must be recognised when selecting the analysis method that WFN and LCA perspectives differ in their general scopes.

References

Allen, R.G., Pereira, L.S., Raes, D., Smith, M. (1998). Crop evapotranspiration: guidelines for computing crop water requirements, FAO Drainage and Irrigation Paper 56, Food and Agriculture Organization, Rome.

Althaus, H.J., Chudacoff, M., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. Life Cycle Inventories of Chemicals. Ecoinvent report No. 8, v2.0 EMPA. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Antón, A., Torrellas, M., Nuñez, M., Sevigné, E., Amores, M., Muñoz, P., Montero, J. (2014). Improvement of agricultural life cycle assessment studies through spatial differentiation and new impact categories: case study on greenhouse tomato production. Environ Sci Technol. 48, 9454-9462.

Antonelli, M., Greco, F. (2013). L'acqua che mangiamo. Cos'è l'acqua virtuale e come la consumiamo. Edizioni Ambiente, Ed.

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., Fusi A., Negri M., Fiala M. (2015). Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J Clean Prod. 86, 49-59.

Bacenetti, J., Lovarelli, D., Fiala, M. (2017). Environmental impact assessment of field mechanisation for a sustainable agriculture. 1st AXEMA EurAgEng Conference on: Intensive and environmentally friendly agriculture: an opportunity for innovation in machinery and systems. Villepinte, France. 27 February 2017.

Bacenetti, J., Lovarelli, D., Fiala, M. (2016a). Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation. Eur J Agron. 79, 107-118.

Bacenetti, J., Bava, L., Zucali, M., Lovarelli, D., Sandrucci, A., Tamburini, A., Fiala, M. (2016b). Anaerobic digestion and milking frequency as mitigation strategies of the environmental burden in the milk production system. Sci Total Environ. 539, 450–459.

Bietresato, M., Calcante, A., Mazzetto, F. (2015). A neural network approach for indirectly estimating farm tractors engine performances. Fuel. 143, 144–154.

Blonk Consultants. (2014). Agri-footprint Description of Data. V 1.0. Retrieved from: www.agri-footprint.com/assets/Agri-Footprint-Part2-Descriptionofdata-Version1.0.pdf.

Boulay, A.M., Hoekstra, A.Y., Vionnet, S. (2013). Complementarities of Water-Focused Life Cycle Assessment and Water Footprint Assessment. Environ Sci Technol. 47(21), 11926–11927.

Boulay, A.M., Bare, J., De Camillis, C., Döll, P., Gassert, F., Gerten, D., Humbert, S., Inaba, A., Itsubo, N., Lemoine, Y., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Schencker, U., Shirakawa, N., Vionnet, S., Worbe, S., Yoshikawa, S., Pfister, S. (2015). Consensus building on the development of a stress-based impact assessment of water consumption: outcome of the expert workshops. Int J Life Cycle Ass. 20, 11367-11374.

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 Ass. 5, 349–357.

Cao, X., Wang, Y., Wu, P., Zhao, X., Wang, J. (2015). An evaluation of the water utilization and grain production of irrigated and rain-fed croplands in China. Sci. Total Environ. 529, 10–20.

Dewesoft catalogue. (2014). Available online at: http://www.dewesoft.com/catalog/#dewesoft-catalog-web_10-2014_v108/page/1-2 [Last access: 15 March 2017].

European Commission-Joint Research Centre - Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook-Recommendations for Life Cycle Impact Assessment in the European context. First edition November 2011. EUR 24571 EN. Luxemburg. Publications Office of the European Union.

European Directive 97/68/EC. On emissions from non-road mobile machinery. In: Official Journal of the European Communities 1998. L 59: 1.

Fellmeth, P. (2003). ISO11783 a Standardized Tractor – Implement Interface. iCC 2003 CAN in Automation, 8–13.

Finnveden, G., Moberg, Å. (2005). Environmental systems analysis tools – an overview. J Clean Prod. 13, 1165-1173.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M. (2007). Overview and Methodology. Ecoinvent Report 1 v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland. Retrieved from: www.ecoinvent.org.

Fusi A., Bacenetti J., González-García S., Vercesi A., Bocchi S., Fiala M. (2014a). Environmental profile of paddy rice cultivation with different straw management. Sci Total Environ. 494–495, 119–128.

Fusi, A., Guidetti, R., Benedetto, G. (2014b). Delving into the environmental aspect of a Sardinian white wine: From partial to total life cycle assessment. Sci Total Environ. 472, 989-1000.

García Morillo, J., Rodríguez Díaz, J.A., Camacho, E., Montesinos, P. (2015). Linking water footprint accounting with irrigation management in high value crops. J Clean Prod. 87, 594–602.

González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L. (2012). Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J Clean Prod. 43, 71-83.

Grisso, R., Perumpral, J.V., Roberson, G., Pitman, R. (2014). Predicting tractor diesel fuel consumption. Virginia Polytechnic Institute and State University. Available online at: http://www.pubs.ext.vt.edu/content/dam/pubs_ext_vt_edu/442/442-073/442-073_pdf.pdf

Hoekstra, A.Y. (2010). The water footprint: water in the supply chain. Water, (93), 12–13.

Hoekstra, A.Y., Hung, P.Q. (2002). Virtual Water Trade. A quantification of virtual water flows between nations in relation to international trade. International Expert Meeting on Virtual Water Trade. 12(11), 1–244.

Ingrao, C., Bacenetti, J., Bezama, A., Blok, V., Geldermann, J., Goglio, P., Koukios, E.G., Lindner, M., Nemecek, T., Siracusa, V., Zabaniotou, A., Huisingh, D. (2016). Agricultural and forest biomass for food, materials and energy: bio-economy as the cornerstone to cleaner production and more sustainable consumption patterns for accelerating the transition towards equitable, sustainable, post fossil-carbon societies. J Clean Prod.117, 4-6.

IPCC, Agriculture, Forestry and Other Land Use. (2006). In: Eggleston et al., IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme.

ISO 14040 series. (2006). Environmental management – Life Cycle Assessment – Requirements and guidelines. International Organization for Standardization.

ISO 14046. (2014). Environmental Management – Water Footprint – Principles, Requirements and Guidelines.

ISO 8178-4. (1996). Reciprocating internal combustion engines – exhaust emission measurements – part 4: Test Cycles for Different Engine Applications. International Organisation of Stadardisation. Geneva, Switzerland.

ISO 11898. (2003). Organization for Standardization. Standards catalogue – ISO/TC 22/SC 3 – electrical and electronic equipment.

Jahns, G., Forster, K-J., Hellickson, M. (1990). Computer simulation of diesel engine performance. Transactions of the ASAE. 764-770.

Janulevičius, A., Juostas, A., Pupinis, G. (2013). Engine performance during tractor operational period. Energ Convers Manage. 68, 11-19.

Janulevičius, A., Juostas, A., Čipliene, A. (2016). Estimation of carbon-oxide emissions of tractors during operation and correlation with the not-to-exceed zone. Biosyst Eng. 147, 117-129.

Janulevičius, A., Juostas, A., Čipliene, A. (2017). Nitrogen-oxide emissions from diesel-engine farm tractors during real-life cycles and their correlation with the not-to-exceed operating zones. Biosyst Eng. 161, 93-105.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J., 2007. Life cycle inventories of bioenergy. Ecoinvent report No. 17. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Lacour, S., Burgun, C., Perilhon, C., Descombes, G., Doyen, V. (2014). A model to assess tractor operational efficiency from bench test data. J Terramechanics. 54, 1-18.

Larsson, G., Hansson, P-A. (2011). Environmental impact of catalytic converters and particle filters for agricultural tractors determined by life cycle assessment. Biosyst Eng. 109, 15-21.

Lazzari, M., Mazzetto, F. (2005). Prontuario di meccanica agraria e meccanizzazione. Ed. REDA.

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 biogas from anaerobic mono-digestion. J Clean Prod. 72, 23-34.

Lindgren, M. (2004). Engine Exhaust Gas Emissions from Non-road Mobile Machinery. Ph.D. Thesis, Department of Biometry and Engineering, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Lindgren, M. (2005). A transient fuel consumption model for non-road mobile machinery. Biosystems Engineering, 91, 139-147.

Lindgren, M., Hansson, P-A. (2002). Effect of engine control strategies and transmission characteristics on the exhaust gas emissions from and agricultural tractor. Biosyst Eng. 83(1), 55-65.

Lindgren, M., Hansson, P-A. (2004). Effect of transient conditions on exhaust emissions from two non-road diesel engines. Biosyst Eng. 87(1), 57-66.

Lindvall, E.E., Törnquist, S., Enghag, O., Lundström, E. (2015). Biogasdrift i arbetsmaskiner. Slutrapport av regeringsuppdrag. Rapport 2015:23.

Lovarelli, D., Bacenetti, J. (2017). Bridging the gap between reliable data collection and the environmental impact for mechanised field operations. Biosyst Eng. 160, 109-123.

Lovarelli, D., Bacenetti, J., Fiala, M. (2016a). Life cycle inventories of agricultural machinery operations: a new tool. J Agr Eng. XLVII, 40-53.

Lovarelli, D., Bacenetti, J., Fiala, M. (2016b). Water Footprint of crop productions: A review. Sci Total Environ. 548-549, 236–251.

Lovarelli, D., Ingrao, C., Fiala, M., Bacenetti, J. (2016c). Beyond the Water Footprint: a new framework proposal to assess freshwater environmental impact and consumption. J Clean Prod. xx, 1-11.

Lovarelli, D., Bacenetti, J., Fiala, M. (2017). Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. J Clean Prod. 140, 479-491.

Mantoam, E.J., Romanelli, T.L., Gimenez, L.M. (2016). Energy demand and greenhouse gases emissions in the life cycle of tractors. Biosyst Eng. 151, 158-170.

Marx, S.E., Luck, J.D., Hoy, R.M., Pitla, S.K., Blankenship, E.E., Darr, M.J. (2015). Validation of machine CAN bus J1939 fuel rate accuracy using Nebraska Tractor Test Laboratory fuel rate data. Comput Electron Agric. 118, 179–185.

Mekonnen, M.M., Hoekstra, A.Y. (2011). The green, blue and grey water footprint of crops and derived crop products. Hydrol Earth Syst Sci. 15, 1577–1600.

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 Bioenerg. 63, 321-329.

Negri, M., Bacenetti, J., Manfredini, A., Lovarelli, D., Maggiore, T.M., Fiala, M., Bocchi, S. (2014b). Evaluation of methane production from maize silage by harvest of different plant portions. Biomass Bioenerg. 67, 339-346.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C. (2015). Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Eur J Agron. 65, 40-51.

Nemecek, T., Ledgard, S. (2016). Modelling farm and field emissions in LCA of farming systems: the case of dairy farming. In: Proceedings of 10th International Conference on Life Cycle Assessment of Food 2016, Dublin. UCD, 1135-1144.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard, R., Halberg, N. (2003). LCA Food Database. www.lcafood.dk.

Niero, M., Ingvordsen, C. H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M. F., Hauschild, M. Z., Jørgensen, R. B. (2015). Eco-efficient production of spring barley in a changed climate: A Life Cycle Assessment including primary data from future climate scenarios. Agricultural Systems, 136, 46–60.

Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds.). (2015). Life Cycle Assessment in the Agri-food Sector. Case Studies, Methodological Issues and Best Practices. Springer. pp. 415.

Notarnicola, B., Sala, S., Anton, A., McLaren S.J., Saouter, E., Sonesson, U. (2017). The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. J Clean Prod. 140, 399-409.

Perozzi, D., Mattetti, M., Molari, G., Sereni, E. (2016). Methodology to analyse farm tractor idling time. Biosyst Eng. 148, 81–89.

Pfister, S., Boulay, A.M. (2017). Understanding the LCA and ISO water footprint: A response to Hoekstra (2016) "A critique on the water-scarcity weighted water footprint in LCA". Ecol Indic. 72, 352-359.

Pfister, S., Koehler, A., Hellweg, S. (2009). Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ Sci Technol. 43(11), 4098–104.

Pitla, S.K., Luck, J.D., Werner, J., Lin, N., Shearer, S.A. (2016). In-field fuel use and load states of agricultural field machinery. Comput Electron Agric. 121, 290–300.

Reboul C. (1964). Temps des travaux et jours disponibles en agriculture. Official document of CIOSTA Institute. Economie rurale. 61 pp. 50-80.

Renzulli, P.A., Bacenetti, J., Benedetto, G., Fusi, A., Ioppolo, G., Niero, M., Proto, M., Salomone, R., Sica, D., Supino, S. (2015). Life Cycle Assessment in the Cereal and Derived Products Sector. In: Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds). Life Cycle Assessment in the Agri-food Sector. Case Studies, Methodological Issues and Best Practices. Springer International Publishing, Switzerland. Pp. 185-249.

Ridoutt, B.G., Pfister, S. (2010). A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change. 20(1), 113–120.

Sala, S., Anton, A., McLaren, S.J., Notarnicola, B., Saouter, E., Sonesson, U. (2017). In quest of reducing the environmental impacts of food production and consumption. J Clean Prod. 140, 387–398.

Scarpare, F.V., Hernandes, T.A.D., Ruiz-Corrêa, S.T., Kolln, O.T., Gava, G.J.D.C., dos Santos, L.N.S., Victoria, R.L. (2016). Sugarcane water footprint under different management practices in Brazil: Tietê/Jacaré watershed assessment. J. Clean. Prod. 112, 4576–4584.

Schäffeler, U., Keller, M. (2008). Non-road fuel consumption and pollutant emissions. Study for the period from 1980 to 2020. p. 171. Bern.

Schmidt Rivera X.C, Bacenetti J., Fusi A., Niero M. (2017). The influence of fertiliser and pesticide emissions model on life cycle assessment of agricultural products: The case of Danish and Italian barley. Sci Total Environ. In press

Speckmann, H., Jahns, G. (1999). Development and application of an agricultural BUS for data transfer. Comput Electron Agric. 23, 219-237.

Suprem, A., Mahalik, N., Kim, K. (2013). A review on application of technology systems, standards and interfaces for agriculture and food sector. Computer Standards & Interfaces, 35(4), 355–364.

Testo® catalogue. (2016). Available online at: http://www.testo350.com/downloads/350/literature-manuals/1-%20testo-350-brochure.pdf [Last access: 15 June 2017].

Thomassen, M.A., Dalgaard, R., Heijungs, R., de Boer, I. (2008). Attributional and consequential LCA of milk production. Int J Life Cycle Assess. 13, 339-349.

Weidema, B.P., Bauer. C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C.O., Wernet, G. (2013). Overview and methodology. Data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1(v3). St. Gallen: The Ecoinvent Centre.

WULCA. (2017). http://www.wulca-waterlca.org/index.html [Accessed May 2017]

Yoo, S.-H., Choi, J.-Y., Lee, S.-H., Kim, T. (2013). Estimating water footprint of paddy rice in Korea. Paddy Water Environ. 12 (1), 43–54.

Attached scientific papers

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A new tool for life cycle inventories of agricultural machinery operations

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Abstract

The interest in environmental assessments about agricultural processes is fast growing and asking for new tools for accurate impact evaluations. The methodology commonly used to go through these studies is the life cycle assessment, of which the inventory phase (life cycle inventory, LCI) is an essential step. For studies focusing on agricultural productions, the completion of LCI is particularly complex: taking into account the pedo-climatic and mechanical operative variability is evidently difficult. However, the prediction of the environmental impact of mechanical operations caused by the agricultural sector is essential to quantify the impact categories for which it is responsible.

A new tool, ENVIAM, was developed to complete LCI to guarantee the availability of local data that describe the mechanical and pedo-climatic conditions occurring in the Po Valley area and widely applicable as well. It calculates mechanical power requests, directly consumed inputs (*i.e.*, fuel, lubricant) and material consumption of a productive system by taking into account soil texture, specific machinery operations and coupling solutions as defined by the user. A subdivision of working time and defined engine load have been considered to calculate fuel consumption; with regard to outputs, exhaust gases emissions from internal combustion engines have been assessed by evaluating the emissive stages of belonging as stated by the EU Directive. A case study was also performed to highlight the differences that occur when an analysis is fulfilled in a context with features different from the average, and resulted in significant variations for the inventory. In more details, a comparison was carried out both with Ecoinvent database and within ENVIAM. With regard to fuel consumption, by changing the soil texture, the analysis showed a range between 64%-184% for sandy and clay soils, respectively, if compared with medium texture ones. With this tool, local contexts defined either as real or as optimised coupling solutions can be investigated to assess their environmental impact.

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Introduction

The recent interest in environmental assessments about agricultural processes is fast growing and asking for tools to make possible accurate sustainability evaluations (Bengoa *et al.,* 2014; Meul *et al.,* 2014; Notarnicola *et al.,* 2015). Being agriculture one of the most impacting sectors (IPCC, 2006), the adoption of methodologies able to guarantee the accurateness and quality of environmental assessments is needed (Jensen *et al.,* 1997; Kerkhof, 2012; Goedkoop *et al.,* 2013). The environmental impact assessment of agro-mechanical operations is an essential component to quantify these responsibilities: they not only concern acidification and eutrophication (*e.g.*, application of fertilisers) but also climate change and resources depletion, as main cause of the use of fossil fuels and raw materials (IPCC, 2006; Notarnicola *et al.,* 2015).

The most useful and emergent methodology to quantify the environmental impacts of several productive sectors is the life cycle assessment (LCA) methodology, characterised by a complete analysis approach made of four phases (ISO standard 14040 series) (ISO, 2006). The phases are: i) goal and scope definition; ii) life cycle inventory (LCI); iii) environmental impact assessment and iv) interpretation. The LCA application to the agricultural sector highlighted some methodological problems because it was developed mainly for industrial processes. In agricultural processes, unlike for industrial ones, inputs and outputs of the system are not always easily measured (Brentrup *et al.,* 2004; Dyer and Desjardins*,* 2003; Ossés de Eicker *et al.,* 2010; Bacenetti *et al.,* 2012; Bengoa *et al.,* 2014)*.* In particular, to obtain primary data (measured), regarding the several emissions sources (in air, water and soil), expensive measures (in terms of money and time) are needed. To overcome this concern, some databases encompassing also the most common agricultural operations are available (*e.g.*, Ecoinvent, the Danish LCA food, the EU and DK input and output database, the agri-footprint database). For agricultural operations, the use of secondary data deriving from these databases is the most applied solution; nevertheless, it is only a partial solution that can introduce uncertainty. The main reason is that, given the database complexity, not all or not always, LCA practitioners are aware of the exact composition of the selected process.

Moreover, each mechanical operation process involved in the databases refers to average conditions and parameters in terms of soil (texture, water content), field shape and slope, distance from the farm and tractors and implement features (*e.g.*, mass, lifespan, annual working time). Therefore, these processes are valid in their related context, but the applicability in different ones should not be taken for granted. When a process from a database is used, it should be modified using inventory data assessed taking into account local pedo-climatic conditions. The availability of a tool that builds a reliable LCI for agricultural operations in the different pedo-climatic contexts, reduces the inaccuracies related to the uncritical use of database processes and can help when an agricultural operation has not been already inserted in databases.

This study relates to the Po Valley area located in Northern Italy, that represents a very important area for Italian agriculture. Common crop production systems consist of cereals (*e.g*., maize, rice and winter crops; ISTAT, 2011) for which usual local mechanical operations are carried out (*e.g*., tillage, crop management, irrigation, drying). By adopting Ecoinvent (Ecoinvent, 2015) in this context, data often result too much simplified and/or totally missing because average European assumptions are not valid (Hansson *et al.,* 2001; Nemecek and Kägi, 2007; Ossés de Eicker *et al.,* 2010; Fiala and Bacenetti, 2012; Bacenetti *et al.,* 2014; Niero *et al.,* 2015; Tendall and Gaillard, 2015). The lack is especially evident when the study focuses on field conditions different from those defined by Ecoinvent. In order to solve these concerns, a tool has been developed to support the completion of an inventory reliable for local conditions.

The aim of this study is the description of the tool, called ENVIAM, *ENVironmental Inventory of Agricultural Machinery operations* that has been developed with the goal of supporting the realisation of a locally reliable inventory about agricultural machinery. The inventory is needed to define inputs and outputs referred to the functional unit to make possible the subsequent phase of the life cycle impact assessment. In addition, the perspicuity of results makes LCA practitioners aware of the inventory data in study.

Materials and methods

ENVIAM description

ENVIAM is the first release of a tool implemented in a MS Office Excel spreadsheet structured to ease the logical steps to follow. It was developed with the goal of assessing a specific inventory valid in the Italian context and applicable in a wide range of different operative conditions and alternatives (*e.g*., machines, soils with different texture).

ENVIAM achieves an accurate quantification of the mechanical parameters (tractor engine power and machinery specific features) and of diesel fuel consumed that most affect agricultural field operations. It performs calculations with usual mechanical knowledge but inputs and outputs of the operations are calculated through refined awareness. Local inputs (*e.g*., fuel, lubricants and other materials) and outputs (*e.g*., exhaust gases emissions) are calculated using both primary (measured) and secondary (context-specific) data. Results can be obtained both for a single operation and for more of them, as the methodology can be retraced for each operation in study. In the end, the results referred to the functional unit (1 ha) or to the total studied field surface can be used to fulfil the LCI phase of an LCA study.

For what concerns the system boundary of each mechanical operation, the analysis includes materials that compose machines, fuel and lubricant and the emission of the main exhaust gases from the combustion in tractor engines.

With regard to the implementation, the user-friendly interface was specifically researched to facilitate the understanding and choice of the parameters. Eleven worksheets were realised and can be discerned in 3 main groups: i) databases (tractor and machinery); ii) support (tables, timing definition, linkage worksheet); iii) calculations (mechanical calculations, fuel and exhaust gases emissions and results).

All worksheets need the user intervention in parameters selection, according to the features characterising the case study. Firstly, two databases of tractors and implements are furnished to have a wide range of labels and models among which to choose the needed machinery. Then, the mechanical operation to carry out must be selected from a list. Currently, the main field operations for crops production typically carried out in the Po Valley are enlisted.

As regards the supporting tables, parameters are given. They must be specifically selected to calculate the requested tractor engine power. Checks are made through tests and the user is directed in all choices.

Finally, ENVIAM calculates fuel consumption (FC), exhaust gases emissions (EM), lubricants and materials consumed during each agricultural machinery operation and results can be related both to the whole operation in study and only to the functional unit (FU), which is 1 ha. By retracing the same steps for each operation, a whole production chain or part of it can be analysed.

Comparisons to investigate the influence on outputs can be made among: i) optimal (best coupling solution of tractor and implement) and other coupling alternatives. Optimal coupling means that the tractor is selected from the database considering the most closed engine power to the one calculated from the tool. Other coupling alternatives can result by the selection of a tractor with too much high (or low) engine power. This choice influences the final outcomes; ii) similar agricultural machines for carrying out a mechanical operation. In this case it means that the same operation is carried out comparing two or more alternative implements; iii) machines adoptable for carrying out alternative operations. Different machines are used for alternative operations in order to analyse the solutions and choose the best alternative.

Databases worksheets

Two databases have been realised; the first is currently composed of 100 tractors (records), classified per label and model and for each of them, mechanical features are enlisted besides (fields). It was fulfilled employing the official Organisation for Economic Co-operation and Development (OECD) reports, Code 2, approved in 2010-2012.

The second database currently lists more than 400 implements (records), on the basis of the operations commonly carried out in the Po Valley for crops production *e.g*., maize, winter cereals). The fields structure is similar to the former database and is reported in Figure 1: label and model, typology of power absorbed [*e.g*., power take off (PTO) or no PTO, towed or carried implement], absorbed engine power, working width and depth (whether necessary), hopper volume (for seed, chemicals and harvested grains), load capacity (for organic and mineral spreaders and for trailers), mass composition (steel, glass, rubber, *etc*.), total mass and engine features specific for self-propelled machinery.

Support worksheets

In the first support worksheet, a wide number of tables are given, distinguishing among different soil types or among agricultural machinery operations. Information is included for adherence and rolling coefficients, forward speed, power surplus coefficient, physical lifespan of tractors, implements and tires, engine load and specific soil resistance. The user must select one of the available values per table, according to the selected operation and to the specific field features.

A second worksheet is related to timings sub-division defined by Reboul (1964; *CIOSTA - Comité International d'Organisation Scientifique du Travail en Agriculture*). In this section, all the working time components defined in the literature (*e.g*., effective work, turns, arrangements, fillings) are reported and the user must measure them on field to fill in the table. Because field shape and size lead to different working time - and LCI -, it is an essential data to measure on field. The measured data are used to compose the total working time (T; h) of the machinery operation (Eq. 1). However, not all of them are compulsory. For instance, during rotary harrowing, neither filling nor emptying time is needed; therefore the input value is 0.

$$T = \sum T_i = TEF + TAV + TAS + TAC + TPL + TME + TMI + TRE + TPH + TRI \qquad (1)$$

where:
T_i (h) is the single temporal component.
TEF is effective working time, TAV turns at headlands, TAS fillings/emptying, TAC field maintenance, TPL implement arrangement on field, TME and TMI avoidable and unavoidable downtime (respectively), TRE worker rest, TPH implement arrangement on farm, TRI transfers.

Calculation worksheets

They are needed for the quantification of:

- Inputs and outputs for each timing component. In particular, they are: i) fuel consumption; and ii) exhaust gases emissions (CO, CO_2, NO_X, HC and PM). With regard to emissions, it must be underlined that the model quantifies the main ones, but that many other gases are emitted, due both to the production of the machines and to that of fuel and lubricant. This section is strictly dependent on tractor engine power, which is the main variable to be quantified. It is calculated according to the selected implement, to traction force and to power requirement. According to the resulting engine power request, the tractor is chosen from the database among those with a similar power to the calculated (Figure 2). In more details, the tractors' database is fulfilled with a wide number of tractors that cover a wide range of engine power (between 30.6-314.1 kW); therefore, the database has enough options to find in any case a tractor with a similar engine power to the calculated request. Indeed, when a similar engine power is not selected, results are affected as follows (Figure 3): when a lower engine power is chosen, the tests in support of engine power request calculation (adherence, lifting capability and longitudinal stability) result negative, meaning that the tractor cannot afford the working conditions; on the opposite, when a too much high engine power is chosen, results are negatively affected because the tractor works with a much lower engine load than it should in optimal conditions and, as a consequence, fuel consumption and exhaust gases emissions are subjected to change. Different outputs from ENVIAM result in different inputs for the environmental impacts assessment through LCA software.
- Inputs and outputs that depend on other variables (time, lifespan,

Table 1. Minimum and maximum engine load values attributed to each working time (*i*- timing).

Working time	λ % Min	λ % Max
Effective time	50	100
Turns time	15	40
Refilling/emptying	0	2
On-field maintenance	0	2
Assume the arrangement setting	0	2
Avoidable downtime	0	2
Unavoidable downtime	0	2
Rest time	0	2
On-farm maintenance	0	2
Transfer time	30	100

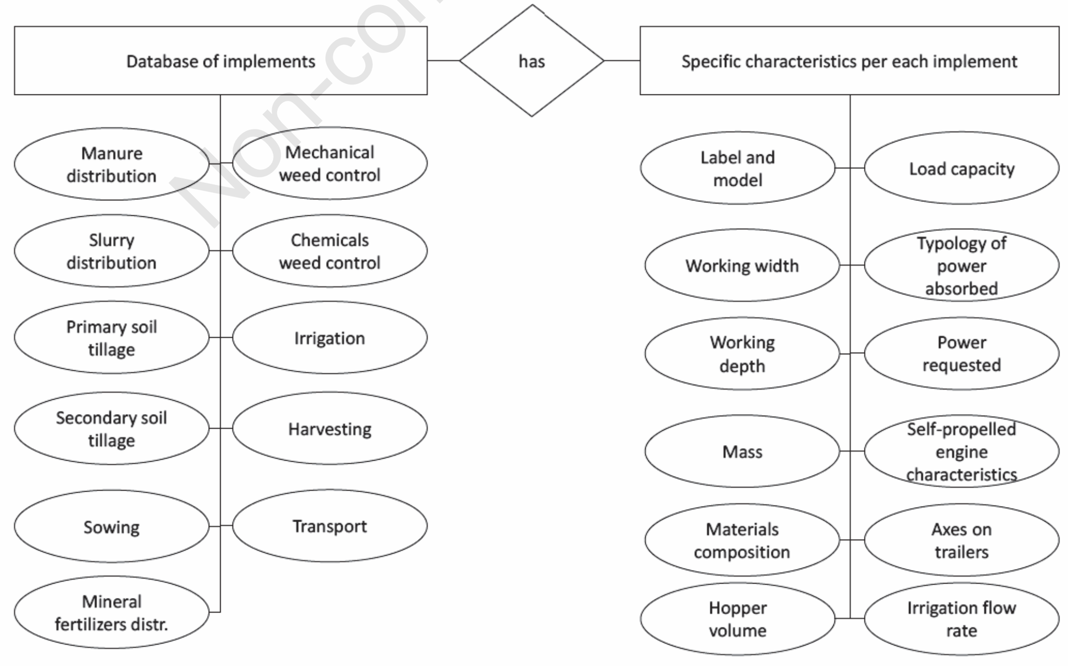

Figure 1. Implements database includes agricultural machinery operations and each implement has records for specific technical characteristics.

masses, *etc*.) are: i) materials consumption; ii) lubricant consumption; iii) input products for the fulfilment of operations (*e.g*., organic and mineral fertilisers, irrigation water). ENVIAM calculates inputs and outputs expressed per field. However, they are also reported to the FU: materials consumption ($kg \cdot ha^{-1}$), fuel and lubricant consumption ($kg \cdot ha^{-1}$) and exhaust gases emissions ($g \cdot ha^{-1}$).

Methodology definition

Engine load (λ)

The engine load (λ; %) is calculated as in Eq. 2:

$$\lambda = \frac{Pm}{Pm_{MAX}} \quad (2)$$

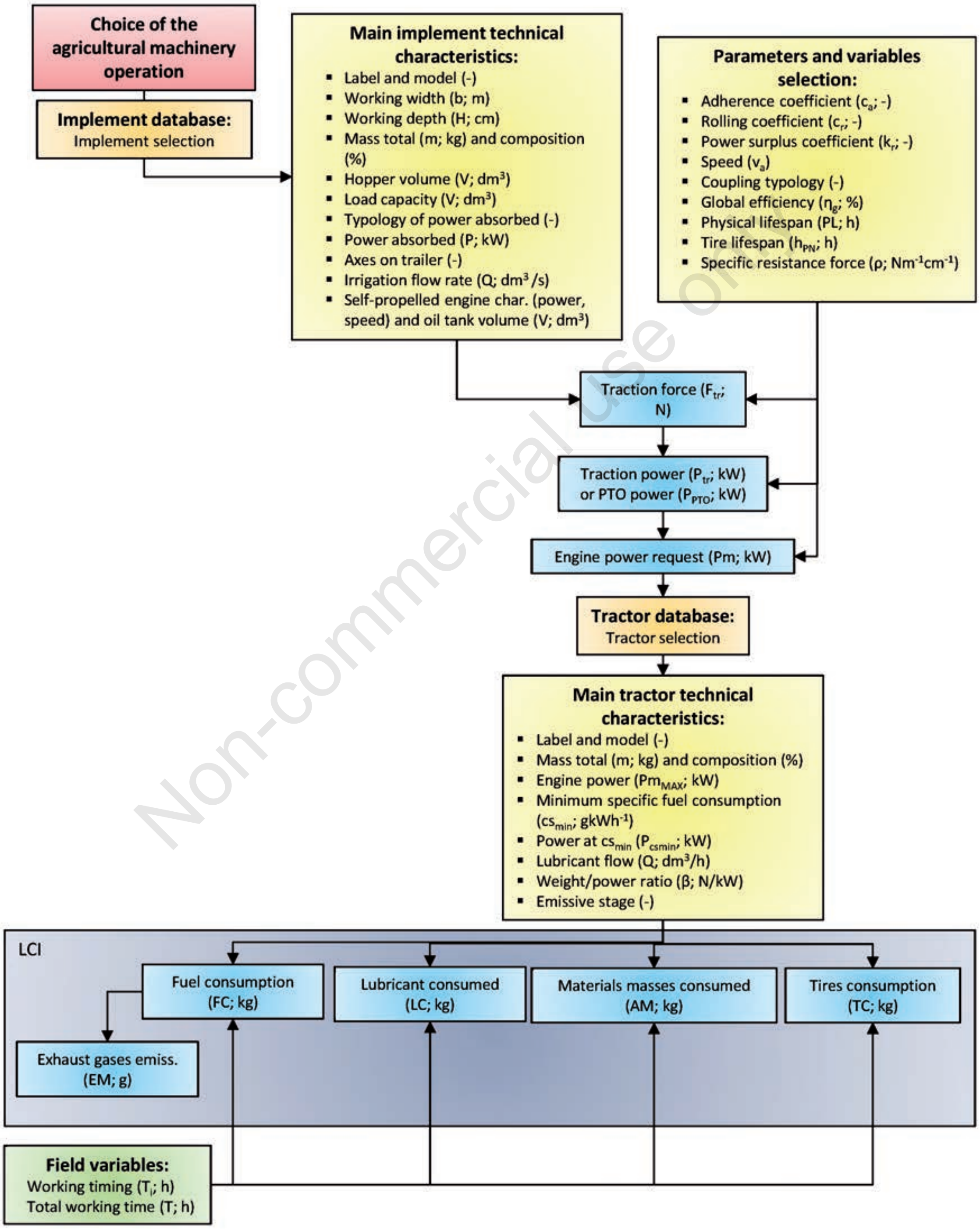

Figure 2. Representation of the steps: from the choice of the agricultural machinery operation to the achievement of results (ENVIAM outputs and life cycle inventory inputs).

where:
Pm (kW) is the engine power absorbed by the implement;
Pm_{MAX} (kW) is the tractor engine power.

According to literature, most of tractors run at 50-70% of Pm_{MAX} during their whole life cycle (Janulevičius *et al.,* 2013a, 2013b; Kim *et al.,* 2013; Lacour *et al.,* 2014). However, λ significantly varies during the operation and strongly influences FC.

Therefore, in ENVIAM, each operation was built composing fractions of working time (Reboul, 1964) in which different average λ occur. Table 1 shows the λ range that was assumed for all the timing components (*i-* timings) and it is recommended for optimal (or nearly optimal) coupling between tractor and implement; however, it varies according to coupling and to the operation. Usually, while effectively working on field the λ is high, but it is not during the whole operation, since λ can be much different during other activities that compose the working time (*e.g.*, during turns, maintenance and arrangement settings, *etc*.). Low λ can occur during turns (the field shape affects the number of turns and, as a consequence, the total turning time characterised by that λ) and refilling/emptying (*e.g.*, manure spreading, pesticides spraying). In these cases, the tractor is on but not working (engine at idle), therefore 0% is advised. About 2% is attributed to those conditions in which only low power is used for the execution of the operation (*e.g.*, hydraulic power). Higher values are attributed to more power-requesting operations.

Fuel consumption

According to literature, there are several methods to calculate FC (Grisso *et al.,* 2004; Lazzari and Mazzetto, 2005; Serrano *et al.,* 2007; Janulevičius *et al.,* 2013a; Sørensen *et al.,* 2014).

ENVIAM takes into account the specific fuel consumption (cs; g·kWh^{-1}), which is defined as the mass of fuel consumed per mechanical energy unit produced (kW·h). cs and the engine load (λ; %) are used to calculate the fuel consumption (FC; kg) applying Lazzari and Mazzetto (2005). The fuel consumption occurring during the *i-* timing (FC_i; kg) is calculated applying Eq. 3:

$$FC_i = \sum cs_i \cdot Pm_{MAX} \cdot \lambda_i \cdot T_i \qquad (3)$$

where:
cs_i (g·kWh^{-1}) is the specific fuel consumption of the *i-* timing;
λ_i (%) is the *i-* engine load.

Total fuel consumption (FC; kg) is the summation of all FC_i that compose the operation (Eq. 4).

$$FC = \sum FC_i \qquad (4)$$

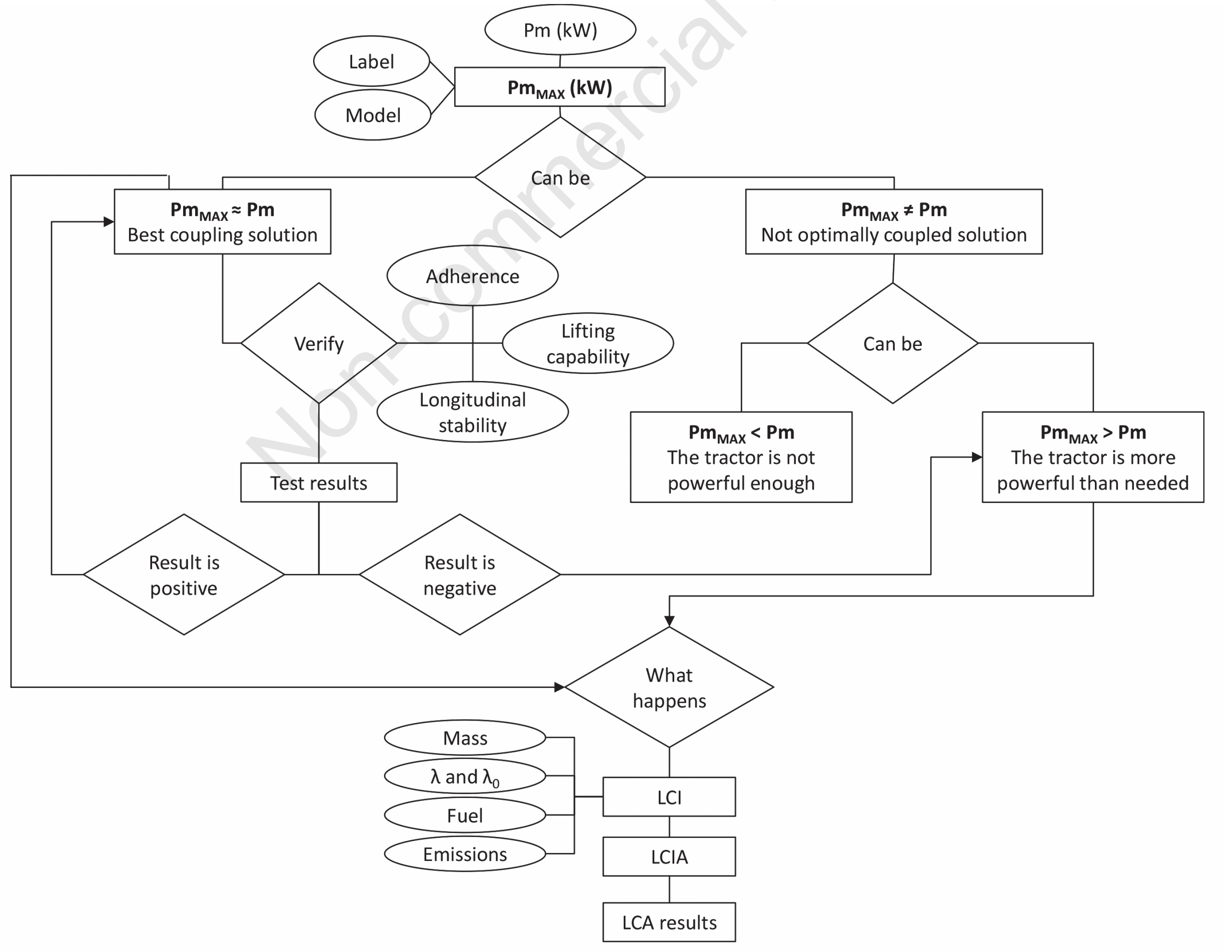

Figure 3. Effects of engine power (Pm_{MAX}) on the coupling.

Exhaust gases emissions

EM are formed during fuel combustion in internal combustion engines and depend on: i) age of the engine; ii) engine power; and iii) engine load with its related dynamic changes.

The year in which the engine was built is crucial, since the emissive limits to which it belongs strongly influence the levels of exhaust gases emissions, as defined by applying the Emissive Standards got from the EU Directive 97/68/EC (and following amending ones: Directive 2010/26/EU, Directive 2010/22/EU) and the ISO standard 8178-4 cycle C1 (European Commission, 1998; ISO, 2007; European Commission, 2010a, 2010b).

Different methods are proposed in literature to assess EM in tractors engines (Hansson *et al.*, 2001; Nemecek and Kägi, 2007; Schäffeler and Keller, 2008; Lindgren *et al.*, 2010; Janulevičius *et al.*, 2013a, 2013b; Kim *et al.*, 2013). However, only Schäffeler and Keller (2008) make available a tool accessible according to the EU Directive and, consequently, to the engine of the specific tractor present in ENVIAM database. The EU Directive dictates engine producers to respect the emissive limits for the main exhaust gases that are CO_2, CO, NO_X, HC and PM. However, during combustion in internal engines, many other gases are produced, but are not quantified in this tool.

In ENVIAM, EM (g) of CO_2, CO, NO_X, HC and PM are calculated according to these authors. Except for CO_2, exhaust gases emissions are quantified as follows:

$$EM_i = \sum EM_{SP} \cdot Pm_{MAX} \cdot \lambda_i \cdot T_i \cdot CF_1 \cdot CF_2 \cdot CF_3 \quad (5)$$

where:
EM_i (g) is the emission of the *i*- timing;
EM_{SP} is the specific limit of each exhaust gas ($g \cdot kWh^{-1}$);
CF_1, CF_2 and CF_3 are correction factors.
By considering different λ_i and T_i, the total emissions of each exhaust gas are calculated (Eq. 6).

$$EM = \sum EM_i \quad (6)$$

As concerns CO_2 from diesel engines, 3.150 g_{CO2}/g_{FUEL} is the factor employed (Schäffeler and Keller, 2008). Once the European Stage of engine belonging is inserted, according to the year of engine production, the proper emission value is picked out. Table 2 shows a numerical example with a tractor that shows different emissions at different working timings.

Amount of machinery and materials

ENVIAM evaluates tractors and implements material consumption (AM; kg) during the operation:

$$AM = \frac{m}{PL} \cdot T \quad (7)$$

where:
m (kg) is the mass (tractor/implement);
PL (h) is the physical lifespan.

PL is the maximum amount of time the machine can work and depends on the typology of work and on the materials needed and used. Average values (Bodria *et al.*, 2006) are suggested but the user can insert different ones according to his study.

In literature, there is an evident lack of data about agricultural machinery composition and manufacturers do not even make these data available. The only accessible are given by Ecoinvent v3 and by few manufacturers' environmental declarations of different machines (*e.g.*, Volvo: http://www.volvoce.com/constructionequipment/corporate/en-gb/environment/publications/Pages/publications.aspx). As regards tractors (AM_{TR}; kg), materials masses consumed were calculated considering tractor masses as defined by OECD Reports (2010-2012) and tractor lifespans by Bodria *et al.* (2006); their subdivision was established as reported by Nemecek *et al.* (2011). For implements (AM_{OM}; kg), several and significant differences among machines for different operations were evident, therefore AM_{OM} was calculated according to information obtained through interviews with experts.

Materials distinction is applied to those tractor materials that are part of the machine for the whole lifespan (partial wear); those continuously consumed are calculated separately (*e.g.*, lubricants, tire sets).

Lubricant consumption

Lubricants substitution in tractors is made according to maintenance schedules, which suggest to substitute engine lubricants on average every 300-400 h of work (T_r; h), whereas in other tractor components the time is longer (gearbox, hydraulic lift, PTO, transmission components).

Tractor lubricant consumption (LC; kg) was calculated with (Eq. 8).

$$LC = \frac{V}{T_r} \cdot \gamma \cdot T \quad (8)$$

where:
V (m^3) is the lubricant volume;
T_r (h) is the retention time (time during which the lubricant remains exploitable in engine/other components);
γ ($kg \cdot m^{-3}$) is the lubricant density.

Moreover, also agricultural machinery LC was reported, as assumed valid for appropriate maintenance schedules and resulting from interviews with experts.

Tire sets consumption

No data were found in literature about tire sets real lifespan except

Table 2. Example of tractor (Pm_{MAX} = 96.4 kW) belonging to emissive Stage 2 that shows at different timing components a different engine load and consequently different exhaust gases emissions for a ploughing operation.

Timings (Ti; h)		Engine load (λ; %)	CO emiss. (EM; g · ha⁻¹)	HC emiss. (EM; g · ha⁻¹)	NO_X emiss. (EM; g · ha⁻¹)	PM emiss. (EM; g · ha⁻¹)	CO_2 emiss. (EM; g · ha⁻¹)
TEF	1.2 h	73%	194.98	45.22	676.53	37.25	60,877.63
TAV	0.12 h	30%	8.02	1.86	27.82	1.53	4852.87
TAC	0.04 h	1%	0.09	0.02	0.31	0.02	103.72
TPH	0.08 h	1%	0.18	0.04	0.62	0.03	207.43
TRI	0.06 h	40%	5.35	1.24	18.55	1.02	2640.07

TEF, effective time; TAV, turns time; TAC, on-field maintenance; TPH, on-farm maintenance; TRI, transfer time.

for Ecoinvent database (Nemecek and Kägi, 2007), which only showed a single average value assumed valid for all tractors and conditions (2500 h).

To compensate for the lack, tractor tire sets lifespan (h_{PN}; h) was assumed from interviews with experts according to the prevailing tractor working activity (Table 3). Attributing different lifespan values permitted to distribute rubbed-off rubber during an adequate time set.

To calculate tire sets mass (m_{PN}; kg), a coefficient k_{PN} (0.0975 $kg_{PN}\cdot kg_{TR}^{-1}$; Nemecek and Kägi, 2007) was employed. It is an empiric ratio between tires mass and tractor mass, validated through measurements on-site as well.

The rubber composing the tire (TC; kg) was calculated with Eq. 9 and distributed along tractor lifespan.

$$TC = \frac{m_{PN}}{h_{PN}} \cdot \frac{1}{k_h} \cdot T \tag{9}$$

where:
m_{PN} (kg) is the tire sets mass;
h_{PN} (h) is tire sets lifespan;
k_h (%) is the ratio between tire sets lifespan and tractor lifespan.

Coupling calculations

Mechanical operations were distinguished in order to take into account the specific power requests. The distinction concerns:

- The typology of implement coupled, so implements are: i) towed (T); or ii) carried (P).
- The operations, which are generally distinguished in: i) primary and secondary soil tillage; ii) crop management; iii) stationary; and iv) transport.

The power absorbed by the implement (Pm; kW) depends on the tractor global efficiency (η_g; %) that characterises the operation; values for η_g are obtained from Bodria *et al.* (2006) and depend on power dissipations that vary between 2WD and 4WD and among the typologies of power request.

As concerns these last, the distinction is among: i) stating (F): the work is carried out in a stationary position (*e.g.*, irrigation, hydraulic woodcutter); ii) towing (T): the implement is only towed by the tractor (*e.g.*, transport); iii) towing and PTO (T+PTO): the implement needs both traction force and PTO power (*e.g.*, organic fertilisers spreading, hay baling); iv) carrying (P): the implement is carried and only traction force is needed (*e.g.*, ploughing, fixed teeth harrowing); v) carrying and PTO (P+PTO): the implement is carried and PTO power is needed (*e.g.*, rotary harrowing, sowing, pesticides spraying, mineral fertilisers spreading).

With regard to both primary and secondary soil tillage operations, required engine power differs as function of soil texture as well; the specific soil resistance (ρ; $N\cdot m^{-1}\cdot cm^{-1}$) values, specific per soil texture and per machinery operation, were taken from literature (Bodria *et al.*, 2006). Similarly, adherence and rolling coefficients originate from Bodria *et al.* (2006). Adherence coefficient is needed in adherence test and rolling coefficient in traction force calculation.

To calculate Pm:

$$Pm = \frac{\mathrm{Ftr} \cdot v_a}{3600 \cdot \eta_g} \tag{10}$$

where:
F_{tr} (N) is traction force - calculated by the tool;
v_a ($km\cdot h^{-1}$) is average forward speed;
η_g (-) is tractor global efficiency - defined by the user in the parameters section.

To calculate Pm for PTO power operations:

$$Pm = \frac{P_{PTO}}{\eta_g} \tag{11}$$

where:
P_{PTO} (kW) is the specific power demanded from the PTO available in the implements' database. Once Pm is calculated, power surplus coefficient (k_r; % - range: 1.05-1.30·Pm) is taken into account, with the purpose of considering potential higher power requests if hard working conditions occur. k_r is selected from the specific table.

Tests

After the tractor selection, tests are made to verify the adequateness of the choice as shown in Figure 4 as well:

- Adherence test (Fiala, 2001; Bodria *et al.*, 2006): to verify that the tractor can overcome external resistance forces. Parameters are already available in ENVIAM: i) tractor mass (m_{TR}; kg); ii) tractor weight/power ratio (β; $N\cdot kW^{-1}$); iii) adherence coefficient (c_a). Verify whether adherence (Ad; N) - calculated by multiplying adherence weight (G_a; N) and adherence coefficient (c_a) - is higher than F_{tr} (N). G_a for 2WD tractors is the weight on the rear axle, whereas for 4WD tractors is the total weight. Whether $Ad < F_{tr}$, ballasting (G_z; kg) is compulsory.

$$G_z = \frac{(F_{tr} - Ad)}{g \cdot c_a} \tag{12}$$

However, if too much ballast is necessary ($G_z \geq 0.3\cdot m_{TR}$) a more powerful (and heavier) tractor is advised.

- Lifting test (ASABE, 1997): for carried implements, to verify that the tractor hydraulic lift is capable of lifting the implement. This test concerns: i) maximum lifted weight (Gs_{MAX}; N); ii) implement mass (m_{OM}; kg); iii) length of parallels on three point hitch till tractor rear axle (br; m); iv) implement length till its centre of gravity (bs; m); v) reference distance b_{rif} (equal to 0.610 m).

$$Gs_{MAX} > \frac{m_{OM}\cdot g\cdot (bs+br)}{(0.610+br)} \tag{13}$$

br and bs were already inserted in ENVIAM, however, the user can use specific values. This is especially valid for bs, as it differs according to implement length. As for br, the value 0.80 m was selected, being valid for a wide range of tractors.

Table 3. Lifespan of tire sets according to the prevailing tractor activity.

Prevailing field operation	Tire sets lifespan (h) Min	Max
Primary soil tillage	4000	5000
Secondary soil tillage	5000	6000
Crop management	6500	8000
Transport	2000	3000
Mixed work	3000	7000

- Longitudinal stability test (D.lgs. 81/08, attachment 5; Italian Regulation, 2008): to verify that the tractor will not overturn longitudinally. Needed data already available in ENVIAM are: i) tractor and implement masses (m_{TR}; kg and m_{OM}; kg); ii) tractor pitch distance (bp; m); iii) length of parallels on three point hitch till tractor rear axle (br; m); iv) implement length till its centre of gravity (bs; m). Italian law D.lgs 81/08, attachment 5 states that 20% of the weight of the tractor plus the worker (assumed standard weight equal to 75 kg) considering bp, br and bs must be higher than the weight of the implement (Italian Regulation, 2008):

$$m_{OM} < \frac{0.2 \cdot (m_{TR} + m_{OM}) \cdot bp}{br + bs} \qquad (14)$$

When this condition is not verified, tractor mass is not suitable to such an implement and either ballasting or another tractor or implement must be selected.

Output worksheet

The output worksheet is divided in two parts. The first includes mechanical data of tractor, implement and operation as consequence of coupling. Data selected in the other worksheets are picked up and for the major part of them, the user plays a basic role: whether the selected value is correct, the test results positive (cell in green background); where no range is reported, the user inserts a value that can be either the same as the calculated or a different one. In addition, a column is devoted to checks. To complete the analysis and if required for the operation, the user can include input materials (IN; kg - *e.g.*, amounts of fertilisers, pesticides, seed, irrigation water, *etc*.); however this is not compulsory. The second part is composed of the results of the agricultural machinery employment. Therefore, it is reported the consumed/emitted amount (total and related to 1 ha) of: i) fuel (FC; kg and kg·ha^{-1}); ii) lubricant (LC; kg and kg·ha^{-1}); iii) tractor and implement (AM; kg and kg·ha^{-1}) both the total and the internal material composition; iv) tire set rubber (TC; g and g·ha^{-1}); v) input products (IN; kg and kg·ha^{-1}); vi) emissions into air due to exhaust gases (EM; g CO_2, g CO, g NO_X, g HC, g PM and g·ha^{-1}).

Results

Goal and structure of the case study

In order to explain how ENVIAM works and the achievable results, a case study concerning ploughing operation was realised. In the following sections, two comparisons are described. First, an explanation of the outputs got from ENVIAM is reported. In more details, the outputs were calculated for three cases in which the type of soil was variable, while implement, working time and all the mechanical parameters not influenced by the soil were the same. The first comparison was made between ENVIAM ploughing process related to a medium texture soil and the same process present in Ecoinvent v3. This analysis aimed at evaluating whether, in the same working conditions, the two tools made feasible the achievement of similar results. The second comparison was made within ENVIAM, analysing the consequences of soil variability on the assessment of mechanical choices (*e.g.*, tractor engine power) and on the consumption of inputs and emission of combustion exhaust gases.

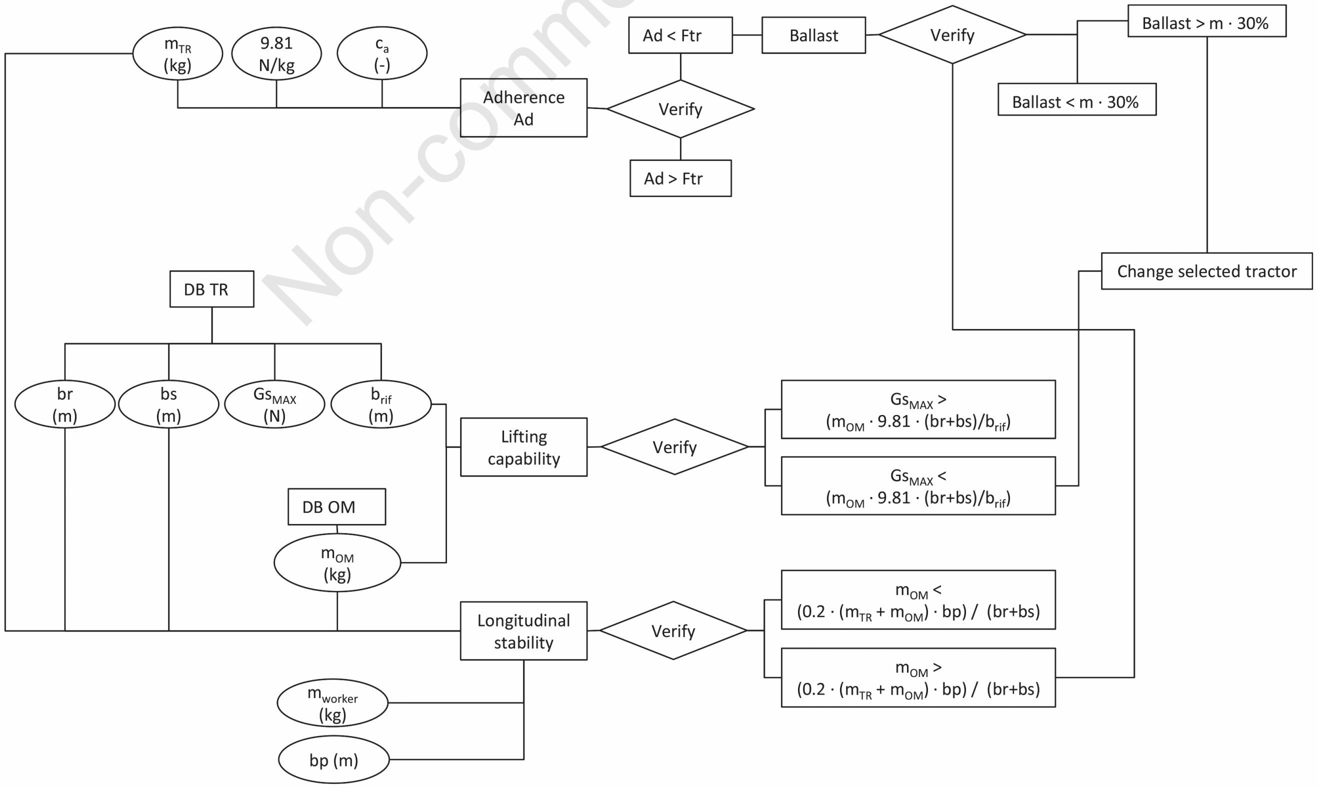

Figure 4. Tests carried out to verify adequateness of the selected tractor. Tests are adherence, lifting capability and longitudinal stability.

Description of ENVIAM outputs

With regard to ENVIAM, a commercial three ploughshares mouldboard plough, mass 1600 kg, was selected from the database to carry out the operation. The soil texture was considered variable, therefore three conditions were assessed: i) sandy soil (SA); ii) medium texture soil (MT); iii) clay soil (CL). For each soil condition, soil resistance (ρ) values were for SA $\rho = 300\ N\cdot m^{-1}\cdot cm^{-1}$, for MT $r = 550\ N\cdot m^{-1}\cdot cm^{-1}$ and for CL $\rho = 1100\ N\cdot m^{-1}\cdot cm^{-1}$. Consequently, power required to carry out ploughing differed and different tractors were needed. For all the three cases, the selected parameters are reported in Table 4.

A tractor with similar Pm_{MAX} was selected from the database verifying adherence (whether needed quantifying ballasting), lifting and stability. With regard to working time composition and engine load, a subdivision is reported in Table 5. The working time was measured on three fields with different soil texture worked by the same operator and the same implement, but with different tractors. Figure 5 reports the followed logical steps. The total time was the same for all the three cases: the effective working time was the same, since working width and speed were constant. As for the other timing components (TAV, TAC, TPH and TIR), time requirements were measured on field and resulted analogous, since the operator was the same and the field shape was similar, in order to assess comparable working conditions. Similarly, engine load was calculated during the effective working time (λ_{TEF}) and differed in each case according to Pm and Pm_{MAX}. On the other components of working time, λ was assumed the same in all three cases.

The total working time was $T = 1.61\ h\cdot ha^{-1}$ and Figure 6 shows time subdivision. Table 6 reports the results of FC, EM, amounts of lubricant, tire and machines consumed (for both tractors and plough) of the three cases evaluated with ENVIAM.

Comparison with Ecoinvent process

In order to compare results of ENVIAM with those of Ecoinvent database, it must be considered that Ecoinvent only considers an average value for medium texture soils (Figure 7).

Therefore, a comparison with ENVIAM could be done only in this condition. The FC in medium texture conditions calculated from ENVIAM was $25.7\ kg\cdot ha^{-1}$ and was similar to that from Ecoinvent ($26.1\ kg\cdot ha^{-1}$), with a difference lower than 2%. Even if in average conditions the variability between the two data sources was small, it can be explained by the calculation method. Fuel consumption was quantified taking into account the engine load in each timing, the specific fuel consumption of the selected tractor and an effective fieldwork capacity defined according to the operative conditions.

With regard to exhaust gases emissions, stronger differences were evident. In particular, for CO, NO_X and CO_2 the difference from ENVIAM was 23%, 180% and 101%, respectively; HC and PM were not comparable to Ecoinvent data. Nevertheless, ENVIAM adopts the recent European Stage emissive standards; therefore, the normative limits expressed in the EU Directive are respected.

Since no consideration could be made on lubricants and tire sets, as ENVIAM and Ecoinvent do not have the same aggregation of results, only amount of machines consumed was compared. This last, however, was relevant, being 218% and 170% higher in Ecoinvent than in ENVIAM. The effective duration of the operation and machines lifespan mainly explained these results.

Comparison with different soil textures

In order to examine the differences occurring by varying soil texture, the medium texture ploughing operation resulting from ENVIAM was compared with the one in sandy and clay soils. Selected options are reported in Table 7. Soil type influences Pm_{MAX} needed to carry out the operation, and, consequently, FC and EM. In particular,

Table 4. Selected input parameters for the case study.

Parameter	Symbol	Value
Average speed	v_a	$6.0\ km\cdot h^{-1}$
Tractor global efficiency	ηg	56%
Working width	L	1.35 m
Working depth	H	0.35 m
Power surplus coefficient	k_r	20%
Tractor lifespan	PL_{TR}	12,000 h
Implement lifespan	PL_{OM}	2000 h
Tire sets lifespan	h_{PN}	4000 h

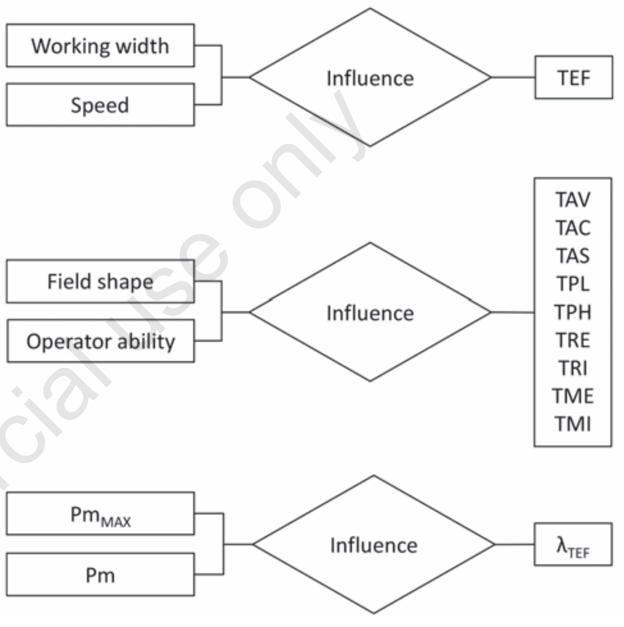

Figure 5. Sharing of the influence of mechanical and local parameters on working time and engine load (λ). TEF, effective time; TAV, turns time; TAC, on-field maintenance; TAS, refilling/emptying; TPL, assume the arrangement setting; TPH, on-farm maintenance; TRE, rest time; TRI, transfer time; TME, avoidable downtime; TMI, unavoidable downtime.

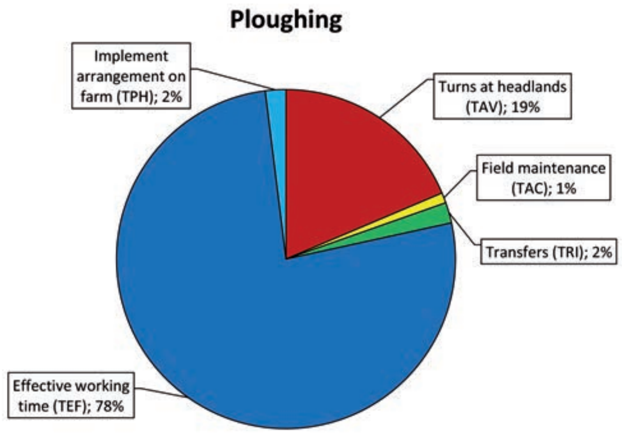

Figure 6. Time subdivision.

FC for ploughing on a sandy soil resulted equal to 65% of the one occurring on a medium texture soil, while was equal to 187% for a clay soil. Exhaust gases emissions showed a similar behaviour as well, highlighting the lowest emissions on sandy soils and the highest on clay soils. The release of CO, HC, NO_X and PM ranged between 45-46% for sandy soils when compared with medium texture ones. CO_2, however, had a similar behaviour to FC in all soil conditions, being directly influenced by fuel consumed. On a clay soil, exhaust emissions from engine tractors during ploughing were 138%, 198%, 120% and 198%, respectively for CO, HC, NO_X and PM of medium texture case. With regard to the amounts of lubricants, tires and materials consumed, the differences occurring among the three tractors (the plough was the same in the three cases), were due to Pm_{MAX} and to the selected tractor masses (3552 kg, 5380 kg and 11,430 kg, respectively for tractors employed on SA, MT and CL). In addition, comparing FC results with the average one from Ecoinvent database results are 64% and 184%, respectively in sandy and clay conditions. Moreover, if not only the soil, but also the implement was varied, stronger differences could have taken place. As an example, with a 5 ploughshares mouldboard plough, total working time would have reduced to 1.29 $h \cdot ha^{-1}$, being wider the working width. FC in sandy, medium texture and clay soils would vary between 79-113%; EM as

Table 5. Subdivision of time and engine load composing case study.

	Plough operation										
Timing subdivision	TEF*	TAV	TAS	TAC	TPL	TME	TMI	TRE	TPH	TRI	Total
Ti ($h \cdot ha^{-1}$)	1.23	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.03	1.61
λ (%)	73[a] 80[b] 79[c]	30	0	2	0	0	0	0	2	40	-

TEF, effective time; TAV, turns time; TAS, refilling/emptying; TAC, on-field maintenance; TPL, assume the arrangement setting; TME, avoidable downtime; TMI, unavoidable downtime; TRE, rest time; TPH, on-farm maintenance; TRI, transfer time. *λ during TEF (λ_{TEF}) varies depending on required power (Pm; kW). Being λ quantified as function of Pm_{MAX} and Pm, the differences in λ_{TEF} are justified by the lower/higher Pm and coupled tractor (different Pm_{MAX}). [a]λ_{TEF} in SA (Pm is 43 kW); [b]λ_{TEF} in MT (Pm is 77 kW); [c]λ_{TEF} in CL (Pm is 155 kW).

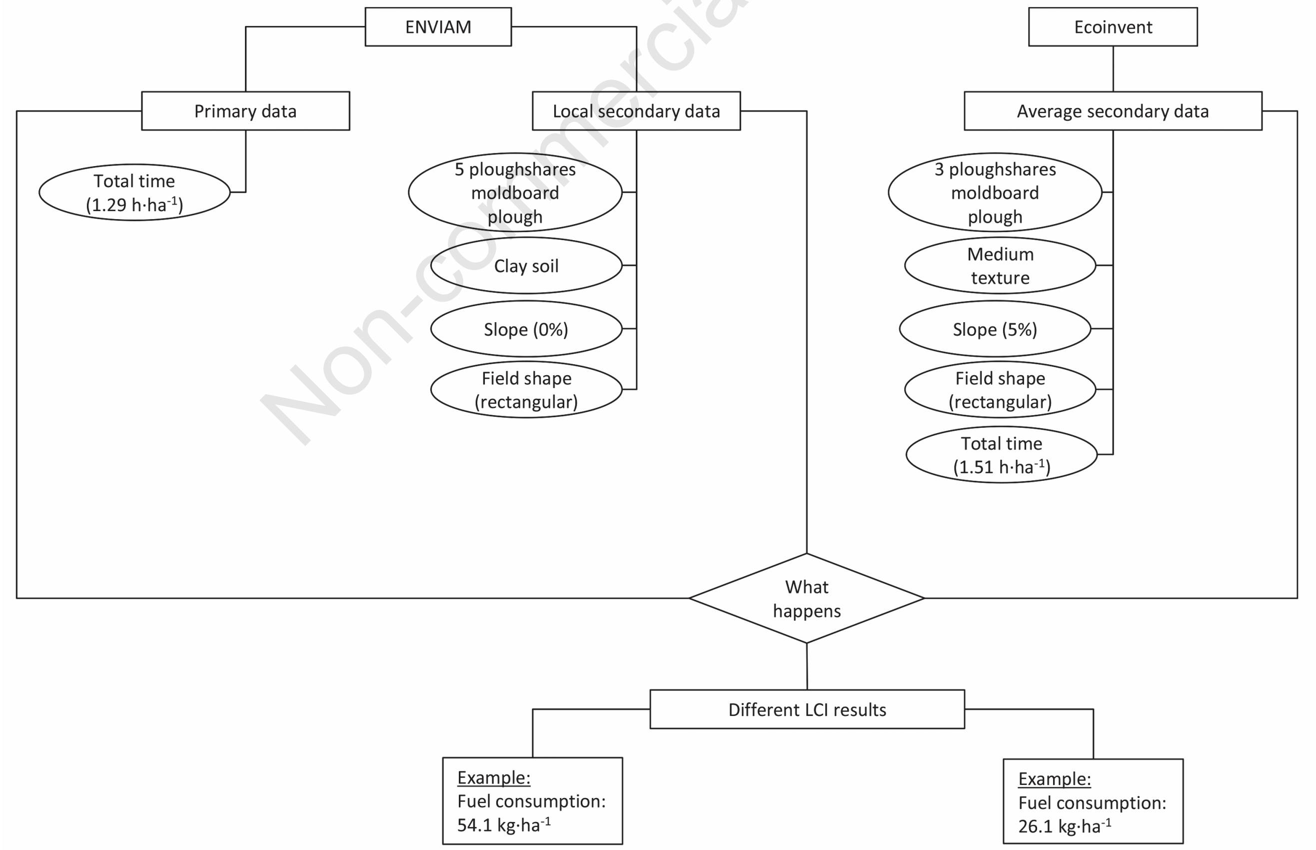

Figure 7. Data employment in ENVIAM and Ecoinvent including an example of the results about fuel consumption for a ploughing operation that is carried out considering local (ENVIAM) and average conditions (Ecoinvent).

well, ranging between 64-143% for all exhaust gases considered, when compared with the ones with three ploughshares. In addition, since more powerful tractors would be needed, also higher amounts of materials, tires and lubricants would be consumed compared to the three-ploughshares, varying between 117%-145% for AM, 73%-147% for LC and 117%-166% for TC.

Discussion

The development of ENVIAM tool brought to raise the awareness that a locally reliable instrument can be significantly helpful to realise a complete and trustworthy LCI. All calculations concerning tractor-implement coupling, tractor's engine power, implement's absorbed power, traction force and slipping are already available in literature. Researchers implemented many models in former years (Lazzari and Mazzetto, 1996; Rotz *et al.,* 1983; Siemens *et al.,* 1990; Haffar and Khoury, 1992; Søgaard and Sørensen, 1996; de Toro and Hansson, 2004). However, ENVIAM uses this typology of well-known calculations to develop a system working with the input databases available. In addition, goal of models developed in the 80s and 90s was the assessment of the economic cost of a coupling decision. On the opposite, goal of ENVIAM is the estimate of locally reliable inventory data used for subsequent environmental analyses of agricultural machinery operations. The calculation of inputs and outputs is obtained from the distinction of the operation in working timings, to each of which, a duration and a specific engine load are attributed. Therefore, the tool takes advantage of a well-known topic, to develop a useful calculation for nowadays-environmental issues. The case study highlighted that, comparing Ecoinvent ploughing process with ENVIAM one in the same soil conditions, low differences (<2%) resulted about fuel consumption. However, they were not negligible with regard to other values, such as for CO and NO_X emissions (23% and 180%, respectively), as well as for tractor and plough consumption (2.2 and 1.7 times higher for Ecoinvent for tractor and plough, respectively). With regard to the comparison carried out within ENVIAM, several differences were highlighted considering the variables taken into account. In particular, the increase in soil resistance forces when the soil was sandy, of medium texture and clay, respectively, caused an increase in tractor engine power requirement, whose satisfaction involved an increase in fuel consumptions and exhaust gases emissions as well as in the materials consumed (bigger masses). About fuel consumption and exhaust gases emissions, the analysis of engine power and load made achievable the reach of higher accurateness. For emissions in particular, the use of the EU

Table 6. Results of fuel consumption, exhaust gases emissions, lubricant, tire and material consumption of the three tractors and plough obtained from the comparison among sandy, medium texture and clay soils in ENVIAM.

Parameters Soil type	SA	ENVIAM MT	CL	
Tractor	Case IH JX 90	Landini Landpower 135 T3	Massey Ferguson 8650	Plough
Pm_{MAX} (kW)	58.8	96.3	196.8	-
FC ($kg \cdot ha^{-1}$)	16.7	25.7	48.0	-
CO ($g \cdot ha^{-1}$)	423.5	914.5	1265.0	-
HC ($g \cdot ha^{-1}$)	16.5	35.6	70.3	-
NO_X ($g \cdot ha^{-1}$)	264.3	570.8	683.5	-
PM ($g \cdot ha^{-1}$)	2.0	4.4	8.7	-
CO_2 ($g \cdot ha^{-1}$)	52,492.3	80,815.4	151,286.2	-
Lubricant ($kg \cdot ha^{-1}$)	1.37	1.14	1.43	0.07
Tire sets ($g \cdot ha^{-1}$)	17.0	67.5	126.5	0.0
TR and OM, total ($kg \cdot ha^{-1}$)	0.48	0.71	1.51	1.27

SA, sandy soil; MT, medium texture soil; CL, clay soil; FC, fuel consumption; TR, tractor; OM, implement.

Table 7. Results of the options selected in ENVIAM for analysing different soil type conditions.

Soil typology	Selected options Sandy	Clay
Tractor	Case IH JX 90	Massey Ferguson 8650
WD	4	4
Pm_{MAX}	58.8 kW	196.8 kW
Pm	42.2 kW	154.7 kW
cs_{min}	$228 g \cdot kWh^{-1}$	$209\ g \cdot kWh^{-1}$
λ_{TEF}	73%	79%
λ_0	93%	79%
Emissive stage	IIIB	IIIB

Directive limits at different Stage of belonging, as well as specific engine loads at different working timings increases the reliability. Moreover, specific engine power, masses, working time and tractor lifespan mark the accurateness increase of outputs as well, which nowadays represents the main lack in Ecoinvent database. This is a particularly relevant issue for LCA studies that focus on agricultural systems. The reason is that studying a system, characterised by local pedo-climatic variables, but using average data will give misleading results. In fact, when the local features are taken into account, Ecoinvent shows inadequate outcomes. In the case study, the average was too much high for sandy soils and too much low for clay soils. Similar evaluations could also be carried out for analogous operations with wider or tinier implements (*e.g*., 5 ploughshares plough) and for other operations, *e.g*., harrowing, seeding, mineral and organic fertilising, harvesting. Even if this study aimed at describing a tool usable for the LCI phase, it is plausible to assume that the resulting differences are relevant for the assessment of the environmental impacts of agricultural machinery as well. If Ecoinvent was applied, at the end of an LCA study, an improper environmental impact assessment would be entailed. However, focusing on ENVIAM implementation, improvements can still be reached. In particular, the increase of operations in the database and a higher amount of machinery represent the main achievable enhancements, also developing other cultivation systems such as other open-field cultivations (*e.g*., potatoes, tomatoes), haying, orchards, as frequently appear in Italian and European cultivation contexts.

Conclusions

The attention on the environmental issues linked to agricultural machinery has recently increased enormously. Many databases and tools are available for research and commercial users to quantify the impacts on the environment of agricultural processes, but their reliability is poor when different pedo-climatic and mechanical conditions occur. In this study, the objective was to describe a tool developed to have those locally reliable inventory data necessary as inputs for subsequent LCA studies. ENVIAM was implemented using data about tractors and implements from literature, technical documentation and manufacturers. Databases of tractors and implements were inserted with the specific goal of having a wide range of ordinary machines, in order to adapt the software tool to the real farm conditions. Moreover, the completely mechanised field chain can be quantified by retracing the same modules for each operation. To show the effect of ENVIAM use on the inventory data, a comparison between its outputs and those from Ecoinvent v3 was done. It showed that average values present in Ecoinvent are applicable only in similar conditions, whereas when soil and implement have diverse features, the differences are not negligible. In conclusion, ENVIAM results not only in a support for LCIs, but is also a standalone tool. It can be applied to studies in which mechanical aspects are linked to environmental evaluations and in which environment-improvement possibilities must be compared and analysed.

References

ASABE (American Society of Agricultural and Biological Engineers). 1997. ASABE S349 - test procedure for measuring hydraulic lift capacity on agricultural tractors equipped with three-point hitch; published 1998-12-01. ASABE, St. Joseph, MI, USA.

Bacenetti J., Fusi A., Fiala M. 2014. Impatto ambientale delle operazioni agricole di campo: effetto della scelta della macchina operatrice e delle condizioni pedologiche. VIII Convegno Rete Italiana LCA, 19-20 June 2014, Florence, Italy.

Bacenetti J., González-García S., Mena A., Fiala M. 2012. Life cycle assessment: an application to poplar for energy cultivated in Italy. J. Agr. Eng. 43:72-8.

Bengoa X., Rossi V., Humbert S., Nemecek T., Lansche J., Mouron P. 2014. Methodological guidelines for the life cycle inventory of agricultural products. Version 2.0, July 2014. World Food LCA Database. Quantis and Agroscope, Lausanne and Zurich, Switzerland, pp. 1-79.

Bodria L., Pellizzi G., Piccarolo P. 2006. Il trattore e le macchine operatrici. Ed. Edagricole, Milano, Italy.

Brentrup F., Küsters J., Kuhlmann H., Lammel J. 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology. Eur. J. Agron. 20:247-64.

De Toro A., Hansson P.-A. 2004. Analysis of field machinery performance based on daily soil workability status using discrete event simulation or on average workday probability. Agr. Syst. 79:109-29.

Dyer J.A., Desjardins R.L. 2003. Simulated farm fieldwork, energy consumption and related greenhouse gas emissions in Canada. Biosyst. Eng. 85:503-13.

Ecoinvent. 2015. Ecoinvent database. Available from: http://www.ecoinvent.org/database/ Accessed: 27 April 2015.

European Commission. 1998. Directive 97/68/EC of the European Parliament and of the Council of 16 December 1997 on the approximation of the laws of the Member States relating to measures against the emission of gaseous and particulate pollutants from internal combustion engines to be installed in non-road mobile machinery. In: Official Journal, L 59, 27/2/1998, pp 1-86.

European Commission. 2010a. Commission Directive 2010/22/EU of 15 March 2010 amending, for the purposes of their adaptation to technical progress, Council Directives 80/720/EEC, 86/298/EEC, 86/415/EEC and 87/402/EEC and Directives 2000/25/EC and 2003/37/EC of the European Parliament and of the Council relating to the type-approval of agricultural or forestry tractors. In: Official Journal, L 91, 10/4/2010, pp 1-68.

European Commission. 2010b. Commission Directive 2010/26/EU of 31 March 2010 amending Directive 97/68/EC of the European Parliament and of the Council on the approximation of the laws of the Member States relating to measures against the emission of gaseous and particulate pollutants from internal combustion engines to be installed in non-road mobile machinery. In: Official Journal, L 86, 1/4/2010, pp 29-47.

Fiala M. 2001. Esercizi di meccanica agraria. Ed. CUSL, Milano, Italy.

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

Goedkoop M., De Oele M., Leijting J., Ponsioen T., Meijer E. 2013. SimaPro tutorial. Available from: https://www.pre-sustainability.com/download/SimaPro8Tutorial.pdf

Grisso R.D., Kocher M. F., Vaughan D.H. 2004. Predicting tractor fuel consumption. Appl. Engine. Agric. 20:553-61.

Hansson P.A., Lindgren M., Norén O. 2001. A comparison between different methods of calculating average engine emissions for agricultural tractors. J. Agr. Eng. Res. 80:37-43.

Haffar I., Khoury R. 1992. A computer model for field machinery selection under multiple cropping. Comput. Electron. Agric. 7:219-29.

IPCC (Intergovernmental Panel on Climate Change). 2006. Agriculture, forestry and other land use, vol.4. In: S. Eggleston, L. Buendia, K. Miwa K, T. Ngara, K. Tanabe (Eds.), IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global

Environmental Strategies (IGES), Hayama, Japan. Available from: http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_00_Cover.pdf

ISO (International Standardisation Organisation). 2006. ISO 14040:2006. Environmental management -- Life cycle assessment -- Principles and framework. Available from: http://www.iso.org/iso/home/store/catalogue_tc/catalogue_detail.htm?csnumber=37456

ISO (International Standardisation Organisation). 2007. ISO 8178-4:2007. Reciprocating internal combustion engines -- Exhaust emission measurement -- Part 4: Steady-state test cycles for different engine applications. Available from: http://www.iso.org/iso/home/store/catalogue_tc/catalogue_detail.htm?csnumber=42275

ISTAT (Italian Institute of Statistics). 2015. Censimento agricoltura. Available from: http://dati-censimentoagricoltura.istat.it/?lang=it Accessed: 27 April 2015.

Italian Regulation. 2008. Decreto Legislativo 9 aprile 2008, n. 81. Attuazione dell'articolo 1 della legge 3 agosto 2007, n. 123, in materia di tutela della salute e della sicurezza nei luoghi di lavoro. In: G.U. Ser. Gen. no. 101, 30/4/2008 - Suppl. Ord. no. 108.

Janulevičius A., Juostas A., Pupinis G. 2013a. Engine performance during tractor operational period. Energ. Convers Manage. 68:11-19.

Janulevičius A., Juostas A., Pupinis G. 2013b. Tractor's engine performance and emission characteristics in the process of ploughing. Energ. Convers Manage. 75:498-508.

Jensen A.A., Hoffman L., Møller B.T., Schmidt A. 1997. Life cycle assessment - A guide to approaches, experiences and information sources. Environmental Issues Series, no. 6. European Environment Agency, Copenhagen, Denmark.

Kerkhof A. 2012. Life cycle-based sustainability - standards and guidelines. Pré Consultants. Available from: https://www.pre-sustainability.com/download/Life-Cycle-Based-Sustainability-Standards-Guidelines.pdf

Kim Y-J., Chung S-O., Choi C-H. 2013. Effects of gear selection of an agricultural tractor on transmission and PTO load during rotary tillage. Soil Till. Res. 134:90-6.

Lacour S., Burgun C., Perilhon C., Descombes G., Doyen V. 2014. A model to assess tractor operational efficiency from bench test data. J Terramechan. S4:1-18.

Lazzari M., Mazzetto F. 1996. A PC model for selecting multicropping farm machinery systems. Comput. Electron. Agric. 14:43-59.

Lazzari M., Mazzetto F. 2005. Prontuario di meccanica agraria e meccanizzazione. Ed. REDA, Torino, Italy.

Lindgren M., Larsson G., Hansson P.A. 2010. Evaluation of factors influencing emissions from tractors and construction equipment during realistic work operations using diesel and bio-fuels as substitute. Biosys. Eng. 107:123-30.

Meul M., Middelaar C.E., de Boer I.J.M., Van Passel S., Fremaut D., Haesaert G. 2014. Potential of life cycle assessment to support environmental decision making at commercial dairy farms. Agr. Syst. 131:105-15.

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

Nemecek T., Kägi T. 2007. Life cycle inventories of agricultural production systems. Ecoinvent report version 2.0. Editors: 0. Volume: 15. Swiss Centre for LCI, ART. Dübendorf and Zurich, Switzerland.

Niero M., Ingvordsen C.H., Peltonen-Sainio P., Jalli M., Lyngkjær M.F., Hauschild M.Z., Jørgensen R.B. 2015. Eco-efficient production of spring barley in a changed climate: A Life Cycle Assessment including primary data from future climate scenarios. Agric. Syst. 136:46-60.

Notarnicola B., Salomone R., Petti L., Renzulli P.A., Roma R., Cerutti A.K. 2015. Life cycle assessment in the agri-food sector. Springer, Milan, Italy.

OECD (Organisation for Economic Co-operation and Development). Reports Code 2. 2010 – 2012.

Ossés de Eicker M., Hischier R., Kulay L.A., Lehmann M., Zah R., Hurni, H. 2010. The applicability of non-local LCI data for LCA. Environ. Impact Assess. Rev. 30:192-9.

Reboul C., 1964. Temps des travaux et jours disponibles en agriculture. Official document of CIOSTA Institute. Economie Rurale 61:50-80.

Rotz C.A., Muhtar H.A, Black J.R. 1983. A multiple crop machinery selection algorithm. Trans. ASAE 26:1644-9.

Schäffeler U., Keller M. 2008. Non-road fuel consumption and pollutant emissions. Study for the period from 1980 to 2020. Environmental studies no. 0828. Federal Office for the Environment, Bern, Switzerland.

Serrano J.M., Peça J.O., da Silva J.M., Pinheiro A., Carvalho M. 2007. Tractor energy requirements in disc harrow systems. Biosyst. Eng. 98:286-96.

Siemens J., Hamburg K., Tyrrell T. 1990. A farm machinery selection and management program. J. Prod. Agric. 3:212-9.

Søgaard H., Sørensen C.G. 1996. A model for optimal selection of machinery sizes within the farm machinery system. pp 588-596 in 6th Int. Conf. on Computers in Agriculture, American Society of Agricultural Engineers, St. Joseph, MI, USA.

Sørensen C.G., Halberg N., Ousdhoorn F.W., Petersen B.M., Dalgaard R. 2014. Energy inputs and GHG emissions of tillage systems. Biosyst. Eng. 120:2-14.

Tendall D.M., Gaillard G. 2015. Environmental consequences of adaptation to climate change in Swiss agriculture: an analysis at farm level. Agr. Syst. 132:40-51.

Glossary

Engine parameters

Maximum power	Pm_{MAX}	kW
Power absorbed by the implement	Pm^*	kW
Power at the minimum specific fuel consumption	Pm_{csmin}	kW
Minimum specific fuel consumption	cs_{min}	$g \cdot kWh^{-1}$
Engine load	λ	%
Engine load at the minimum specific fuel consumption	λ_0	%

Tractor (TR)

Forward speed	v_a	$km \cdot h^{-1}$
Global efficiency	η_g	%
Traction force	F_{tr}	N
Traction power	P_{tr}	kW
Power at PTO	P_{pto}	kW
Weight/power ratio	β	$N \cdot kW^{-1}$
Adherence weight	G_a	N
Adherence coefficient	c_a	-
Ballasting	Gz	kg
Power surplus coefficient	k_r	-
Pitch distance	bp	m
Length of parallels on three-point hitch till tractor rear axle	br	m
Distance between TR's journal boxes and OM's centre of gravity	bs	m
Mass	m	kg
Maximum lifted weight	Gs_{MAX}	N

Implement (OM)

Working depth	H	cm
Working width	b	m
Rolling coefficient	c_r	-
Effective field capacity	EFC	$ha \cdot h^{-1}$

Field and working time components

Surface	A	ha
Total time for fieldwork operations	T	h
Effective time	TEF	h
Turns time	TAV	h
Filling-emptying time	TAS	h
On-field maintenance time (regulations)	TAC	h
Avoidable downtime (working disorganisation)	TME	h
Unavoidable downtime (sudden breakings)	TMI	h
On-farm preparation time (coupling/ uncoupling)	TPH	h
On-field preparation time (arrange the working layout)	TPL	h
Rest time	TRE	h
Transfer time	TRI	h
Physical lifetime	PL	h

Materials consumption and emissions

Amount of machine (TR and OM)	AM	$kg \cdot ha^{-1}$
Emissions	EM	g
Specific combustion gases emissions	EM_{SP}	$g \cdot kWh^{-1}$
Fuel consumption	FC	$kg \cdot ha^{-1}$
Lubricant retention time	T_r	h
Lubricant consumption	LC	$kg \cdot ha^{-1}$
Tire lifetime	h_{PN}	h
Tire mass	m_{PN}	kg
Ratio between tire and tractor lifetime	k_h	%

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Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage

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ABSTRACT

The interest in environmental assessments about agricultural processes is high and asks for tools for accurate impact evaluations. The methodology commonly used in these studies is the Life Cycle Assessment (LCA), of which the inventory phase (Life Cycle Inventory – LCI) is the essential and most complex step to fulfil, for agricultural productions in particular. The reason is that taking into account local variables such as soil texture and mechanical operative solutions for the agro-mechanical operations is difficult.

The aim of this study was to perform a case study to quantify the environmental impacts through LCA of alternative ploughing solutions and to quantify the differences that occur when an analysis is fulfilled with inventories completed with two different tools. First, when a database furnishes average data (Ecoinvent) and, secondly, when the inventory is completed with a tool that considers local variables. In particular, the used new tool is ENVIAM (ENVironmental Inventory of Agricultural Machinery operations), which was developed to take into account local variables. Finally, a subsequent goal was to quantify the environmental impacts through LCA of alternative ploughing solutions. Using ENVIAM, mouldboard ploughs were compared with slatted ploughs and variables such as the number of ploughshares, the field shape ratio (i.e. the ratio of field length and width considering regular quadrangles) and soil texture differed. Fuel consumption and exhaust gases emissions were calculated as function of working time, engine load and European Standard Emissive Stage. The functional unit was "1 ha tilled in a primary soil tillage operation appropriately and completely carried out" and the International Reference Life Cycle Data System (ILCD) characterization method was used for the impact assessment. The most common implement present in Northern Italy, the 3 ploughshares mouldboard plough, was considered as baseline scenario. When working on medium texture soils, discrepancies with Ecoinvent were not negligible (less than 9% for Climate Change and Ozone Depletion). However, they resulted even 2–3 times higher for Particulate Matter and Mineral and Fossil Resources Depletion. Instead, when soil texture differed, dissimilarities were considerably higher. For example, Climate Change impact category ranged between −46.2% and +108.1% of the identified baseline case (with sandy and clay soils, respectively).

1. Introduction

The assessment of environmental impacts of agricultural activities is spreading as consequence of the attention given to environmental sustainability issues. To carry out these assessments, the methodology to use is the Life Cycle Assessment (LCA) (CEC, 2003), characterized by a complete analysis approach (ISO 14040, 2006). Even though it is a widespread tool, carrying out a LCA can still be complicated because the inventory fulfilment is complex. However, databases to complete inventories are commonly available (e.g., Ecoinvent, the Danish LCA food, the EU and DK input and output database, Ecoinvent, 2015; Nielsen et al., 2003; Jannick et al., 2010). Their limit is the accurateness, which represents an undeniable drawback to truthful environmental assessments (Bengoa et al., 2014; Meul et al., 2014; Notarnicola et al., 2015; Falloon and Betts, 2010). With regard to some production sectors, however, the lacks in databases cause big concern. In absence of both trustful databases as well as primary (measured) data, different tools are necessary for the completion of inventories with acceptable quality (Dyer and Desjardins, 2003; Ossés de Eicker et al., 2010; Bengoa

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et al., 2014). For example, the agricultural sector is responsible for notable environmental impacts (Intergovernmental Panel on Climate Change, IPCC, 2006), such as climate change, acidification, eutrophication, ozone depletion and mineral and fossil resources depletion. In this sector, agro-mechanical operations play a significant role on the environmental effects (Intergovernmental Panel on Climate Change, IPCC, 2006; Hokazono and Hayashi, 2012; Notarnicola et al., 2015; Dace et al., 2015; Keyes et al., 2015): each mechanical operation is characterised by operative choices that are the result of local concerns (soil texture, field shape ratio, declivity, climatic issues such as temperature and rainfall). Their variation affects operational choices that cause, in turn, more or less remarkable environmental impacts. In particular, soil texture and implement dimension influence the farmer operational choices (i.e. selection of an adequate tractor, work organization on farm). These local characteristics are difficult to obtain, to measure or even to implement in a database (Dyer and Desjardins, 2003; Ossés de Eicker et al., 2010; Bengoa et al., 2014), but methodologies able to guarantee accurateness and quality of inventories for agro-mechanical operations are essential for appropriate environmental assessments (Jensen et al., 1997; Kerkhof, 2012; Goedkoop et al., 2013). In fact, with regard to some agricultural activities, the most widely employed database for LCA, which is Ecoinvent (Ecoinvent, 2015), furnishes simplified or totally missing information on some agricultural processes (Tendall and Gaillard, 2015; Niero et al., 2015; Bacenetti et al., 2015a, 2015b; Ingrao et al., 2015). In particular, when the analysis focuses on conditions (e.g., soil texture, field shape ratio, implements) that differ from averages and/or common assumptions, the lack in information is tedious.

In order to have a tool able to support the fulfilment of inventories with respect to local variability, the tool ENVIAM (ENVironmental Inventory of Agricultural Machinery operations) was developed for agricultural machinery operations (Lovarelli et al., in press) and was here employed to fulfil the inventory of a LCA about ploughing. This tool was built by the Department of Agricultural and Environmental Sciences, Production, Landscape, Agroenergy, of the Università degli Studi in Milan. In this study, the inventory completed with ENVIAM was inserted in a LCA software to evaluate the influence of soil texture and operative conditions on the environmental impacts assessment of primary soil tillage. Primary soil tillage is the grouping of those operations (e.g., ploughing, subsoiling, scarifying) carried out to alter the soil state, from a noncultivated to an unrefined one. Among the agro-mechanical operations, primary soil tillage is one of those more deeply affected by soil texture and operative variables (Bacenetti et al., 2014a). Soil tillage was chosen in this study because it is a basic operation and, even if its technical development already reached the maximum, it is one of the most frequent agricultural process hotspots on the environmental point of view (Brentrup et al., 2004; Fusi et al., 2014; Bacenetti et al., 2013b, 2014b; Noya et al., 2015).

The aim of this study is the assessment of the environmental impacts of ploughing operation carried out both with different soil textures (sandy, medium texture and clay soils) and with different machinery (both 3 and 5 ploughshares mouldboard ploughs and 3 ploughshares slatted ploughs) in order to identify and quantify the error made when no local information is used. Moreover, one additional option was the variation in field shape ratio, to analyse whether a different environmental impact could be highlighted in this condition. The study analysed agricultural working conditions typical of the Po Valley in Northern Italy, which is the most important Italian agricultural area for crop production systems (e.g., maize, rice and winter cereals, in particular; ISTAT, 2015).

The novelty of this study is not the assessment of the environmental impacts of ploughing operation, but the quantification of the different environmental impacts of ploughings carried out in diverse local conditions. In particular, it is essential to underline that what emerges from databases is adequate for average working conditions, but is misleading, both in negative and in positive terms, when the database does not have the same assumptions of the study.

The paper is divided in two parts: first, the description of the model ENVIAM is given, while in the second part the model is employed for supporting the fulfilment of the Life Cycle Inventory (LCI) of primary soil tillage operations in order to assess their environmental impacts.

2. Methods

2.1. ENVIAM description

For the most widespread field operations, ENVIAM (Fig. 1) permits to achieve an accurate quantification of (i) the mechanical parameters (tractor engine power and machinery specific features), (ii) diesel consumption, (iii) lubricant consumption (iv) materials consumption and (v) exhaust gases emissions. In more details, the tool permits to couple tractors and implements present in two databases (one for tractors and one for implements) and highlights possible inconsistencies from classical mechanics literature through tests. Afterwards, according to on-field working time collected data, inputs and outputs are calculated for the operation. The usefulness of ENVIAM is due to the possibility of splitting the working time of an operation on fractions and attributing to each of them a suitable engine load. The database selection of a tractor and an implement for a defined operation allows for the quantification of inputs and outputs specific for the local studied conditions. In particular, local pedo-climatic (e.g., soil texture, field shape ratio and size) conditions deeply affect the tractor engine power request. Different tractors (more or less powerful) and implements, determine the inclusion of inventories characterised by features (e.g., masses, lifespan, fuel consumption) that affect LCA results.

Using ENVIAM, the performable comparisons to investigate the influence of the tractor choice on outputs are between:

(i) optimally and not-optimally coupled tractor and implement: the best coupling solution between tractor and implement is calculated in terms of tractor power and its results can be compared with those deriving from other coupling alternatives (e.g., high power compared to the implement need);
(ii) similar implements for carrying out the same operation: for example, mouldboard ploughs compared with slatted ploughs can be used to carry out the same operation, but have a different mechanical and environmental impact.

Eleven worksheets were realised and can be discerned in 3 major groups: (i) databases, (ii) parameters selection and (iii) calculations. A detailed description of the tool and its methodology was reported in Lovarelli et al. (in press).

2.1.1. Databases

Tractors and implements were inserted in two databases fulfilled with the official Organization for Economic Co-operation and Development (OECD) reports referred to Code 2, (2010–2012) and with commercial leaflets for the typical operations of crops production carried out in Northern Italy. The objective was to have brands and models that cover a wide range of machines options.

2.1.2. Parameters selection

Specific tables concerning mechanical parameters (e.g., specific soil resistance, tractor global efficiency, adherence coefficient), working time and a linkage worksheet for user selections were

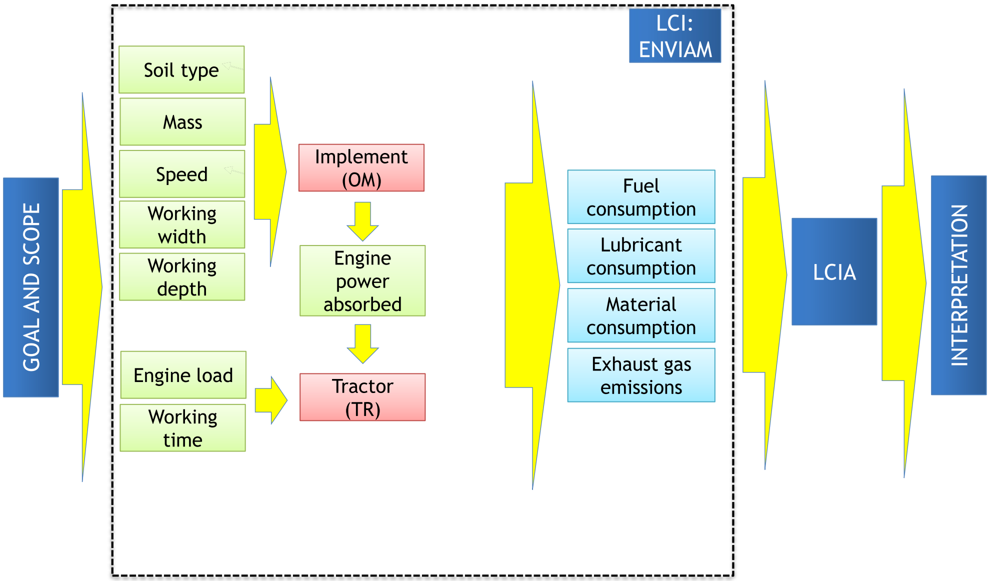

Fig. 1. Specific logical definition of ENSAM and its insertion in a Life Cycle Assessment.

made. In particular, ENVIAM can be applied in a wide range of cultivation contexts because mechanical parameters affected by pedo-climatic conditions (e.g., different soil texture, field shape ratio) show values discriminated among alternative contexts (e.g., soil texture, soil cover, typology of coupling). Users can select different mechanical parameters according to the local conditions. Among the selections and the mechanical parameters, total working time and engine load were essential to be considered in view of ENVIAM scope, as their variation strongly affected the assessments result. Total working time of an operation (T; h) was assessed by summing the single *i*-time components (T_i; h) previously measured on field/farm. Time sub-division was defined according to Bolli and Scotton (1987) including effective working time (TEF), turns at headlands (TAV), filling/emptying (TAS), preparation on farm (TPH), settings on field (TPL), maintenance on field (TAC), rest time (TRE), avoidable downtime (TME), unavoidable downtime (TMI) and transfer farm/field (TIR).

To each of these, a specific engine load (λ_i; %) was connected. λ_i is the ratio between the engine power absorbed by the implement (Pm; kW) in the *i*-time and the maximum engine power of the tractor (Pm_{MAX}; kW). To define λ_i, recent literature was considered (Janulevičius et al., 2013a, 2013b; Kim et al., 2013; Lacour et al., 2014).

2.1.3. Calculations

Mechanical calculations were done to indicate the best tractor-implement coupling and, consequently, to calculate the consumption of:

(i) fuel (FC; $kg \cdot ha^{-1}$),
(ii) lubricant (LC; $kg \cdot ha^{-1}$),
(iii) tractor and implement (AM; $kg \cdot ha^{-1}$),

and the emission to atmosphere of:

(i) exhaust gases (EM; $g \cdot ha^{-1}$ CO_2, $g \cdot ha^{-1}$ CO, $g \cdot ha^{-1}$ NO_X, $g \cdot ha^{-1}$ HC, $g \cdot ha^{-1}$ PM).

2.1.3.1. Fuel consumption (FC).

Fuel consumption (FC; kg) was obtained as the summation of all FC_i.

$$FC = \sum FC_i = \sum cs_i \cdot Pm_{MAX} \cdot \lambda_i \cdot T_i \qquad (1)$$

where:

FC_i (kg): fuel consumption during each *i*-time
cs_i ($g \cdot kWh^{-1}$): specific fuel consumption (Lazzari and Mazzetto, 2005) during each *i*-time
Pm_{MAX} (kW): tractor maximum power
λ_i (%): engine load during each *i*-time
T_i (h): *i*-time.

2.1.3.2. Lubricant consumption (LC).

Lubricant consumption (LC; kg) was calculated as:

$$LC = \frac{V}{T_r} \cdot \gamma \cdot T \qquad (2)$$

where:

V (m^3): lubricant volume consumed in engine, gearbox, hydraulic lift, PTO and transmission components
T_r (h): ordinary turnover period indicated by manufacturer
γ ($kg \cdot m^{-3}$): lubricant density
T (h): total working time of the operation.

2.1.3.3. Tractor and implement material consumption (AM).

For the calculation of the materials consumed during tractor and implement use (AM; kg), Eq. (3.1)–(3.2) were applied.

$$AM_{TR} = \frac{m_{TR}}{PL_{TR}} \cdot T \qquad (3.1)$$

$$AM_{OM} = \frac{m_{OM}}{PL_{OM}} \cdot T \qquad (3.2)$$

where:

m_{TR} (kg): mass of tractors obtained from OECD reports and from the application of Nemecek et al. (2011).

m_{OM} (kg): mass of implements obtained from commercial leaflets, interviews with experts and from the application of Nemecek et al. (2011).
PL_{TR} (h): tractor physical lifespan (Bodria et al., 2006)
PL_{OM} (h): implement physical lifespan (Bodria et al., 2006).

2.1.3.4. Exhaust gases emissions into air (EM). Emissions of exhaust gases (EM; g) are formed during fuel combustion in internal combustion (i.c.) engines and strongly depend on the engine operating mode (i.e. mechanical features). Secondly, they also depend on the year in which the engine was built, because of the steep normative reduction of exhaust gases emissions (emissive Stages, i.e., I, II, IIIA, IIIB, IV). The reference laws were the Emissive Standards of the EU Directive 97/68/EC (and following amending ones: Directive 2010/26/EU, Directive 2010/22/EU) shown in Table 1 and the ISO standard 8178-4 cycle C1.

To assess EM of tractors engines, literature was reviewed (Hansson et al., 2001; Nemecek and Kägi, 2007; Schäffeler and Keller, 2008; Lindgren et al., 2010; Kim et al., 2013; Janulevičius et al., 2013a, 2013b). Schäffeler and Keller (2008) resulted accessible according to the EU Directive and possible to arrange with each tractor engine available in ENVIAM. The EU Directive dictated engine producers to respect the emissive limits for the main exhaust gases (i.e., carbon dioxide, CO_2; carbon monoxide, CO; nitrogen oxides, NO_X; hydrocarbons, HC and particulate matter, PM) however, many other gases are produced during combustion, but not quantified in ENVIAM. In details, except for CO_2 that was calculated employing the factor for diesel engines equal to 3.150 g_{CO2}/g_{FUEL}, CO, NO_X, HC and PM emissions were quantified as follows (Schäffeler and Keller, 2008):

$$EM = \sum EM_i = \sum EM_{SP} \cdot Pm_{MAX} \cdot \lambda_i \cdot T_i \cdot CF_1 \cdot CF_2 \cdot CF_3 \qquad (4)$$

where:

EM_i (g): emission of each exhaust gas occurring in the $i-$ time
EM_{SP} ($g \cdot kWh^{-1}$): specific emission factor of each exhaust gas
CF_1: correction factor for deviation of effective engine load from the standard load on which the emission factor is based
CF_2: correction factor for tractor dynamic use
CF_3: correction factor for tractor wear and tear.

Table 1
Emissive limits of off-road vehicles according to the Stage of belonging of the engine (EU Directive 97/68/EC and amending ones).

Gas	Engine power class (kW)	Emission limits ($g \cdot kWh^{-1}$)				
		I	II	IIIA	IIIB	IV
CO	18–37	–	5.50	5.50	–	–
	37–56	6.50	5.00	5.00	5.00	–
	56–75	6.50	5.00	5.00	5.00	5.00
	75–130	5.00	5.00	5.00	5.00	5.00
	130–560	5.00	3.50	3.50	3.50	3.50
HC	18–37	–	1.50	–	–	–
	37–56	1.30	1.30	–	–	–
	56–75	1.30	1.30	–	0.190	0.190
	75–130	1.30	1.00	–	0.190	0.190
	130–560	1.30	1.00	–	0.190	0.190
NO_X	18–37	–	8.00	–	–	–
	37–56	9.20	7.00	–	–	–
	56–75	9.20	7.00	–	3.30	0.40
	75–130	9.20	6.00	–	3.30	0.40
	130–560	9.20	6.00	–	2.00	0.40
HC + NO_X	18–37	–	–	7.50	–	–
	37–56	–	–	4.70	4.70	–
	56–75	–	–	4.70	–	–
	75–130	–	–	4.00	–	–
	130–560	–	–	4.00	–	–
PM	18–37	–	0.80	0.60	–	–
	37–56	0.85	0.40	0.40	0.025	–
	56–75	0.85	0.40	0.40	0.025	0.025
	75–130	0.70	0.30	0.30	0.025	0.025
	130–560	0.54	0.20	0.20	0.025	0.025

2.2. LCA of primary tillage operations

During the last years, in several studies emerged the role of agricultural machinery operations on the environmental impacts of the complete cropping cycle. Niero et al. (2015) carried out a LCA study on Danish barley. Authors reported an inventory where ploughing covered 43% of fuel consumption of the complete cropping system. Fallahpour et al. (2012) studied Iranian barley and wheat production and stated that fuel consumption during ploughing, planting, spraying and harvesting was the major responsible for global warming. Similar results were obtained from Fedele et al. (2014) on Italian barley and soybean, where ploughing was the operation responsible for the highest fuel consumption. Bacenetti et al. (2013a) carried out a LCA study on maize cultivation in Italy showing impacts per each operation and resulted that mechanical operations have the biggest role on global warming. Concerning a study completed in Italy, Noya et al. (2015) reported that 26% (for both wheat and triticale) and 20% (for maize) of total fuel consumption were used during ploughing. An Italian case study by Bacenetti et al. (2015b) showed the deep impact of primary soil tillage operations caused by fuel consumption. The effect raised mainly on global warming potential and abiotic resources depletion impact categories. Authors reported that, until sowing, 46% of fuel consumption was due to ploughing in maize and triticale cultivation. A detailed inventory of fuel consumption per each mechanical operation for Italian rice production was also reported in Fusi et al. (2014) where ploughing was responsible for the highest fuel consumption (26% of total consumption in the cropping system). Blengini and Busto (2009) stated that Italian rice production was highly mechanised and that an environmental benefit could be achieved with lower mechanisation levels. Similarly, Chiaramonti and Recchia (2010) studied Italian sunflower oil production and showed that fuel consumption during ploughing was 38% of total consumption for mechanised operations.

Having seen that ploughing is a mechanical operation with a considerable responsibility on the environmental concerns during crops cropping cycles, alternative options to carry out the tillage were assessed. In more details, in order to study the effect and the influence of soil texture and of different operative conditions on the environmental impacts of ploughing, ENVIAM was used as LCI data source. Outputs were calculated for 11 cases in which soil type, implement typology and field shape ratio were varied, while mechanical parameters not influenced by the soil were kept constant. In addition to these, the case of ploughing process included in Ecoinvent database was reported. Consequently, a total of 12 cases were analysed. This last was used to quantify the difference between the environmental impacts obtained from a locally reliable inventory that consisted of 11 different conditions and those from a database-completed inventory in which only one condition was available. The goal was to identify the relevance and to quantify the error that occurred when average assumptions did not reflect the study local conditions.

In more details, the research questions can be summarised as follows:

(i) What is the environmental impact of a soil tillage operation?
(ii) Which are the local conditions that have a higher influence on the environmental impacts of soil tillage?
(iii) Are databases for the fulfilment of LCI reliable enough for all working conditions and/or contexts?

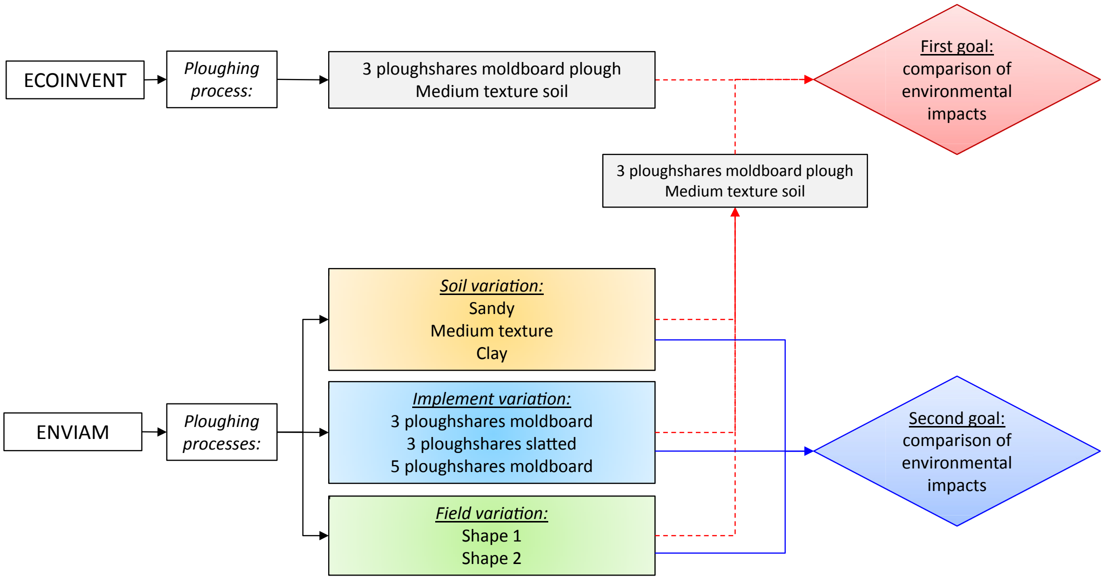

Fig. 2. Goal of the case study.

(iv) Is it always correct to attribute to soil tillage operations the title of "cultivation hotspot process" in the environmental impacts assessment of agricultural activities?

The outcomes of such an analysis can be helpful for farmers, farmer associations, technicians, stakeholders as well as politicians involved in the decision-making of both the processes of cropping systems and the environmental concerns about soil tillage impacts.

2.2.1. Goal and scope

Environmental impacts were calculated for 12 cases. Of them, 11 were inventoried through ENVIAM and 1 by using Ecoinvent ploughing process.

As shown in Fig. 2, the first goal was to make a comparison between the environmental impacts resulting from the analysis of the mouldboard ploughing process related to a medium texture soil using ENVIAM and the same process inventoried through Ecoinvent v3. This analysis aimed at evaluating whether, in the same working conditions, the impacts from the two tools were similar.

The second goal was to make a comparison among the environmental impacts of local conditions occurring during ploughing operations. This Life Cycle Inventory (LCI) was fulfilled only with ENVIAM as data source, analysing the consequences of soil variability, implement selection and field shape ratio on the assessment of mechanical choices (e.g., tractor power) and on the environmental impact assessment. The difference occurring between each of these and Ecoinvent was discussed, in order to reflect on the error occurring when average assumptions do not reflect the local ones.

2.2.2. System description

Assuming the same soil moisture, tractors and implements were selected in order to perform the ploughing with the same final effect on soil preparation.

The cases analysed through ENVIAM were made according to different soil texture, implements and field shape ratios. Concerning soil texture were assessed:

(i) sandy soil (SA),
(ii) medium texture soil (MT), and
(iii) clay soil (CL).

For each of these, ploughs compared were:

(i) 3 ploughshares mouldboard (3M),
(ii) 5 ploughshares mouldboard (5M), and
(iii) 3 ploughshares slatted (3S).[1]

The field shape ratio was considered rectangular (50 m wide, 200 m long) and was the same for these eight cases of ploughing operations compared, which are:

(i) 3 ploughshares mouldboard plough on sandy soil 3M_SA,
(ii) 3 ploughshares mouldboard plough on medium texture soil 3M_MT,
(iii) 3 ploughshares mouldboard plough on clay soil 3M_CL,
(iv) 5 ploughshares mouldboard plough on sandy soil 5M_SA,
(v) 5 ploughshares mouldboard plough on medium texture soil 5M_MT,
(vi) 5 ploughshares mouldboard plough on clay soil 5M_CL,
(vii) 3 ploughshares slatted plough on medium texture soil 3S_MT,
(viii) 3 ploughshares slatted plough on clay soil 3S_CL.

However, a comparison was also carried out in all the three soil texture conditions between 3M and a scenario (3M+) in which the field shape ratio was squared and required, consequently, a higher number of turns at the headlands. In more details:

(i) 50 m wide and 200 m long field for 3M,
(ii) 100 m wide and 100 m long field for 3M+.

All options were assessed according to the optimal coupling solution achieved through calculations in ENVIAM.

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[1] The case with a 3 ploughshares slatted plough (3S) on sandy soil (SA) was not assessed because it is an uncommon solution. Usually, slatted plough is employed only on heavy soils to reduce the traction force. Being the soil sandy, the traction force is already low and, therefore, the slatted plough is not used in real working contexts.

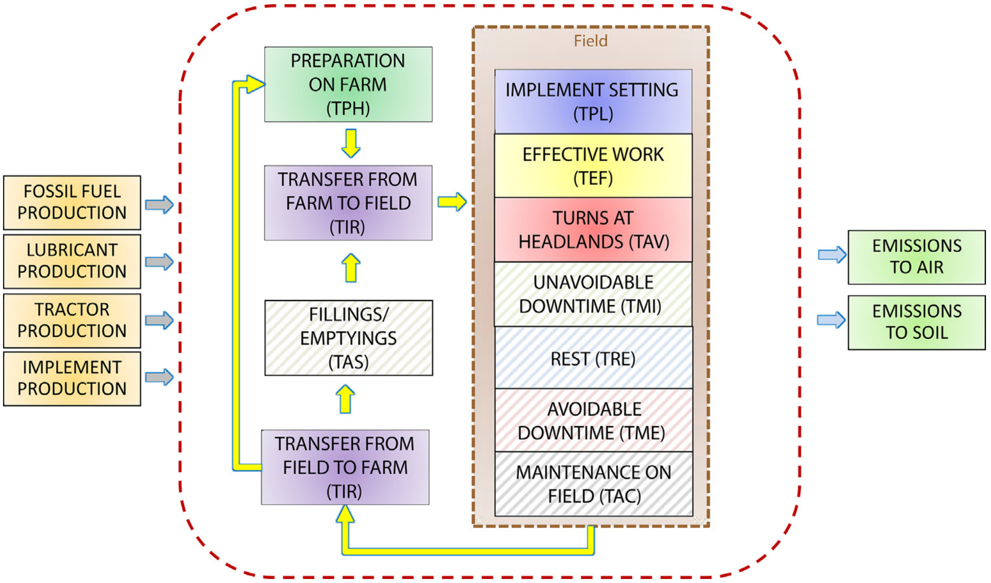

Fig. 3. System boundary. Boxes in the central part of the figure refer to working time that is split in effective work (TEF) and ancillary time components. Ancillary components are: time for tractor and implement preparation on farm – TPH; time due to transfers farm-to-field and vice versa – TIR; time to fill/empty the hopper – TAS; time to install the machines working setting on field – TPL; time for turns at the headlands – TAV, rest time for the worker – TRE, avoidable downtime (bad organisation of workers) – TME, unavoidable downtime (machines breakages) – TMI, maintenance of machines on field (adjustments, clogging) – TAC. Boxes coloured with diagonal lines are meant for ancillary working components that not always are due to take place.

2.2.3. Functional unit and system boundary

The functional unit is an important step of Life Cycle Assessments since it provides the reference to which all other data in the assessment are normalised. With LCA's application to agricultural processes, different functional units (FUs) can be selected. In this study, the goal was to compare different primary soil tillage operations after whose completion the same final effect on the soil was reached. Therefore, the selected FU was "1 ha tilled in a primary soil tillage operation appropriately and completely carried out".

As regard to the system boundary, Fig. 3 shows the system boundary of soil tillage operation split in its working components and the inputs and outputs considered. In more details, raw materials extraction (e.g., fossil fuels and minerals), manufacture (e.g., agricultural machines), use (diesel fuel consumption and derived exhaust gases emissions and tyre abrasion), maintenance and final disposal of machines were included. In addition, the indirect environmental burdens of capital goods (tractors, implements and buildings) were also included.

2.2.4. Life Cycle Inventory

The Life Cycle Inventory (LCI) was fulfilled using both primary and secondary data. Within each implement typology (3M, 5M, 3S), working times were directly measured on field using a digital chronometer. Fields had different soil texture, were worked by the same operator and the same implement. All of them were located at 400 m distance from farm. The total time was calculated by summing all the *i*-time components and resulted being the same for 3M and 3S. In particular, the effective working time was the same, since working width and speed were constant. Regardless from the strict assumption to keep constant speed in all cases, the reason was that to each implement was coupled a tractor characterised by optimal power. As for the other ancillary components (turns at the headlands TAV, maintenance on field TAC, preparation on farm TPH and transfers TIR), the measure on field/farm resulted analogous to each other, since the operator was the same and the field shape ratio was equal (comparable working conditions).

The measured total working time was $T = 1.61\ h \cdot ha^{-1}$ for 3M and 3S, while was $T = 1.29\ h \cdot ha^{-1}$ for 5M. Case 3M+ (square field shape) entailed a higher total working time ($T = 1.71\ h \cdot ha^{-1}$). Table 2 shows time subdivision.

Secondary data derive both from estimates done in ENVIAM and from Ecoinvent database. With regard to those carried out within ENVIAM, were calculated: (i) fuel consumption ($kg \cdot ha^{-1}$), (ii) lubricant consumption ($kg \cdot ha^{-1}$), (iii) tractor material consumption ($kg \cdot ha^{-1}$), (iv) implement material consumption ($kg \cdot ha^{-1}$), (v) exhaust gases emissions ($g \cdot ha^{-1}$).

Literature, OECD reports and commercial leaflets were analysed for their assessment. In particular, specific data of the inventoried tractors and implements were used (e.g., tractor mass, implement mass, tractor power, specific fuel consumption, working width,

Table 2
Subdivision of time composing case study.

Cases	Time subdivision T_i ($h \cdot ha^{-1}$)										
	TEF	TAV	TAS	TAC	TPL	TME	TMI	TR	TPH	TIR	Total
3 mouldboard ploughshares plough (3M)	1.23	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.03	1.61
3 slatted ploughshares plough (3S)	1.23	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.03	1.61
3 mouldboard ploughshares plough, different shape (3M+)	1.33	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.03	1.71
5 mouldboard ploughshares plough (5M)	0.91	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.03	1.29

Notes: For time subdivision, no difference in total working time was highlighted among the soils considered.

Table 3
Subdivision of engine load composing the 11 cases analysed. SA: sandy soil; MT: medium texture soil; CL: clay soil.

Cases			Engine load λ_i (%)									
Soil	Implement	Tractor (Pm_{MAX})	TEF	TAV	TAS	TAC	TPL	TME	TMI	TRE	TPH	TIR
SA	3 mouldboard ploughshares plough (3M)	58.8 kW	73	30	0	2	0	0	0	0	2	40
MT		96.3 kW	80									
CL		196.8 kW	79									
MT	3 slatted ploughshares plough (3S)	85.2 kW	81									
CL		166.4 kW	89									
SA	3 mouldboard ploughshares plough, different shape (3M+)	58.8 kW	73									
MT		96.3 kW	80									
CL		196.8 kW	79									
SA	5 mouldboard ploughshares plough (5M)	90.8 kW	76									
MT		154.7 kW	81									
CL		252.1 kW	70									

etc.). To calculate the necessary variables, local parameters were used as well. Because during primary soil tillage the essential parameter to consider was the resistance opposed by the soil to the operation progress, for each soil condition evaluated (SA, MT and CL) a different specific soil resistance (ρ; $N \cdot m^{-1} \cdot cm^{-1}$) was assumed: for SA $\rho = 300\ N \cdot m^{-1} \cdot cm^{-1}$, for MT $\rho = 550\ N \cdot m^{-1} \cdot cm^{-1}$ and for CL $\rho = 1100\ N \cdot m^{-1} \cdot cm^{-1}$ (Bodria et al., 2006). However, for slatted ploughs, ρ was smaller, since lower resistance forces had to be overcome; in particular, for MT $\rho = 490\ N \cdot m^{-1} \cdot cm^{-1}$ and for CL $\rho = 979\ N \cdot m^{-1} \cdot cm^{-1}$. Moreover, were considered: (i) average speed $v_a = 6.0\ km \cdot h^{-1}$; (ii) tractor global efficiency $\eta_g = 56\%$; (iii) working width L = 1.35 m (L = 2.25 m for 5M); (iv) working depth H = 0.35 m; (v) power surplus coefficient $k_r = 1.2$ (20%); (vii) tractor and implement lifespans PL_{TR} = 12,000 h and PL_{OM} = 2000 h, respectively, and (viii) tyre sets lifespan h_{PN} = 4000 h. Consequently, for each case, power required to carry out ploughing differed and different tractors were needed. The tractor was selected from the database choosing the one characterised by the most similar maximum power to the one required for ploughing (i.e., optimal coupling). Table 3 reports the engine load (λ_i) that was calculated during the effective working time (λ_{TEF}) because it depended on Pm, whereas it was assumed the same for all the analysed cases in the ancillary components. Finally, emissions of exhaust gases were calculated for all tractors. All of them belonged to the emissive Stage IIIB, so that comparable conditions were assured. Table 4 reports the LCI of inputs and outputs for the 11 cases completed through ENVIAM.

Secondary data concerning the production of the different inputs were obtained from Ecoinvent database (Ecoinvent, 2015) (Table 5).

2.2.5. *Life cycle impact assessment*

The characterisation factors reported by the ILCD method were used (Wolf et al., 2012). The following nine impact potentials were evaluated according to the selected method: climate change (CC), ozone depletion (OD), particulate matter (PM), photochemical ozone formation (POF), acidification (AC), freshwater eutrophication (FE), terrestrial eutrophication (TE), marine eutrophication (ME) and mineral, fossil and renewable resource depletion (MFRD).

3. Results

Table 6 reports the environmental impact in absolute values for the FU considered of all the 11 cases evaluated with ENVIAM and the case reported in Ecoinvent. In total, the environmental impact of 12 primary soil tillage operations (ploughing) was evaluated.

The process hotspots are reported in Table 7 and are shown in the Supplementary Material. For each analysed case and for each impact category, the environmental hotspots were distinguished among process emissions, inputs production (tractor, plough, fuel plus lubricant) and shed, expressed as a percentage of the total impact. Process emissions included all emissions to air (related to fuel combustion in the tractor engines) and to soil occurring during the field operation; they resulted mainly responsible for CC (range between 68% and 77%), POF (range between 65% and 83%), AC (range between 55% and 77%), TE (range between 77% and 91%) and ME (range between 77% and 91%). Tractor production was mainly responsible for MFRD (range between 80% and 91%) because minerals (e.g., steel, chromium, copper) and energy used during its production affected in major part this impact category. Machinery production was the process hotspot for FE (range between 42% and

Table 4
LCI of the 11 cases for fuel consumption (FC), lubricant consumption (LC), tractor consumption (AM_{TR}) implement consumption (AM_{OM}) and emissions (EM) fulfilled using ENSAM.

		FC	LC	AM_{TR}	AM_{OM}	EM_{CO}	EM_{HC}	EM_{NOX}	EM_{PM}	EM_{CO2}
		$kg \cdot ha^{-1}$	$kg \cdot ha^{-1}$	$kg \cdot ha^{-1}$	$kg \cdot ha^{-1}$	$kg \cdot ha^{-1}$	$g \cdot ha^{-1}$	$kg \cdot ha^{-1}$	$g \cdot ha^{-1}$	$kg \cdot ha^{-1}$
Cases	3M_SA	16.7	1.3	0.48	1.27	0.42	16.5	0.26	2.0	52.3
	3M_MT	25.7	1.1	0.72	1.27	0.91	35.6	0.57	4.4	80.8
	3M_CL	48.0	1.4	1.51	1.27	1.26	70.3	0.68	8.7	151.2
	3S_MT	25.0	1.6	0.66	1.44	0.83	32.4	0.52	4.0	78.6
	3S_CL	42.1	1.5	1.14	1.44	1.21	67.4	0.65	8.3	132.4
	3M_SA+	17.5	1.4	0.59	1.37	0.43	16.9	0.27	2.1	55.1
	3M_MT+	27.0	1.2	0.77	1.37	0.94	36.6	0.58	4.5	85.0
	3M_CL+	50.4	1.6	1.63	1.37	1.30	72.3	0.70	8.8	158.9
	5M_SA	15.4	0.8	0.64	1.82	0.50	19.7	0.31	2.4	48.4
	5M_MT	29.0	1.7	0.87	1.82	0.71	39.5	0.38	4.8	91.2
	5M_CL	54.1	1.1	1.96	1.82	1.39	77.6	0.75	9.5	170.3

Notes: 3M_SA: 3 mouldboard plough, sandy soil; 3M_MT: 3 mouldboard plough, medium texture; 3M_CL: 3 mouldboard plough, clay soil; 3S_MT: 3 slatted plough, medium texture; 3S_CL: 3 slatted plough, clay soil; 3M_SA+: 3 mouldboard plough, sandy soil, different field shape; 3M_MT+: 3 mouldboard plough, medium texture, different field shape; 3M_CL+: 3 mouldboard plough, clay soil, different field shape; 5M_SA: 5 mouldboard plough, sandy soil; 5M_MT: 5 mouldboard plough, medium texture; 5M_CL: 5 mouldboard plough, clay soil.

Table 5
Ecoinvent unit processes involved in the inventory.

Unit process	Ecoinvent process
Diesel fuel	Diesel, at regional storage/RER U
Lubricant oil	Lubricating oil, at plant/RER U
Tractor	Tractor, production/CH/I U
Operative machine	Agricultural machinery, general, production/CH/I U
Shed	Shed, CH/I U

72%) because it was composed by a higher amount of steel, which is the major responsible for FE. Fuel production was the most impacting on OD (range between 89% and 95%) and PM (range between 13% and 50%) because of the emissions to air related to its production.

The primary soil tillage process (ploughing) carried out with a 3 mouldboard plough on a medium texture soil (3M_MT) was considered as reference for the comparison with the other primary soil tillage processes. This choice was due to the fact that the 3 ploughshares mouldboard plough was the implement commonly used in most farms of the Po Valley. Moreover, the case with medium texture was selected, as it was the condition most spread on the Po Valley, usually considered the best on the agronomic point of view. A second reason was that ploughing operation in Ecoinvent database was assessed on a medium texture soil with a 3 ploughshares mouldboard plough and a comparison between these two data sources was practicable. Fig. 4 shows the results of this relative comparison.

Fig. 5 shows the comparison among the environmental impact of the analysed cases. In each impact category, the case with the highest environmental impact (worst environmental behaviour) was set equal to 100%, while the other cases were all referred to it. This meant that, differently from Fig. 4, the reference was not anymore 3M_MT, but the worst environmentally impacting option. In more details, the two most impacting processes were Ecoinvent ploughing process (ECO) and the 5 ploughshares mouldboard plough on clay soil (5M_CL).

For PM, POF, AC, TE, ME, the highest impacts resulted from Ecoinvent process (ECO), while for the remaining categories (CC, OD, FE, MFRD) the highest impacts were due to the ploughing operation carried out with 5M_CL.

(i) For CC, the lowest and highest impacts were shown for the 5 ploughshares mouldboard plough on sandy soil (5M_SA) and 5 ploughshares mouldboard plough on clay soil (5M_CL), respectively. When considering the 3 ploughshares mouldboard plough on medium texture soil (3M_MT) as reference for all cases, 5M_SA resulted in an impact reduction of −34.0% and 5M_CL showed an impact increase of +108.1%. Considering that for this impact category process emissions were the main responsible, the reasons for the significant discrepancy between the lowest and highest impact could be identified: fuel consumption was one key hotspot. On a sandy soil, being the soil resistance to overcome lower than on a medium texture and clay soil, the selected tractor, in the same working time and specific fuel consumption, had the lowest total fuel consumption. Similarly, exhaust gases emissions increased at the increase of fuel consumption and at the reduction of engine load, and in the same emissive Stage, showed a comparable behaviour to fuel. Lubricant consumption depended on tractor specific features and, in particular, on the total working time. Moreover, with a lower working time, also the consumed machinery mass was lower, which also explained the linked different environmental impact for process emissions.

(ii) For OD, 5 ploughshares mouldboard plough on sandy soil (5M_SA) had the highest impact reduction, which was equal to −36.8% of the reference 3M_MT. The highest impact increase occurred for the 5 ploughshares mouldboard plough on clay soil (5M_CL) that was 210.1% of 3M_MT. For this impact category, the most important hotspot was the amount of fuel produced that represented the main cause of this huge difference between the best and worst environmental solutions. In particular, these two cases were characterised by the lowest (5M_SA) and highest (5M_CL) fuel consumptions.

(iii) With regard to PM, the 3 ploughshares mouldboard plough on sandy soil (3M_SA) showed again the lowest impact because low exhaust gases were emitted to air. PM impact category showed that process emissions were the main responsible for this category effect. 3M_SAwas equal to −30.5% compared to 3M_MT, while ECO (3 ploughshares mouldboard plough of Ecoinvent process) resulted in an increase of +231.8% compared to 3M_MT. The reason for high emissions in ECO was that Ecoinvent quantified exhaust gases emissions with a different method, not updated to the EU Directive. In fact, all cases fulfilled with ENVIAM belonged to IIIB emissive Stage, characterised by strict emission limits.

(iv) For POF, AC and TE, the lowest environmental impacts resulted from the 3 ploughshares mouldboard plough on sandy soil (3M_SA), being −50.6%, −47.9% and −56.3% of the reference (3M_MT), respectively. For the same categories, the

Table 6
Environmental impact of the soil tillage operations analysed.

Impact category	Unit	Cases											
		3M_MT	3M_SA	3M_CL	3S_MT	3S_CL	3M_SA+	3M_MT+	3M_CL+	5M_SA	5M_MT	5M_CL	ECO
CC	kg CO2 eq	$1.09\cdot10^{-2}$	$8.13\cdot10^{-3}$	$1.98\cdot10^{-2}$	$1.06\cdot10^{-2}$	$1.74\cdot10^{-2}$	$8.60\cdot10^{-3}$	$1.14\cdot10^{-2}$	$2.08\cdot10^{-2}$	$7.17\cdot10^{-3}$	$1.25\cdot10^{-2}$	$2.26\cdot10^{-2}$	$1.18\cdot10^{-2}$
OD	kg CFC-11 eq	$1.57\cdot10^{-9}$	$1.15\cdot10^{-9}$	$2.89\cdot10^{-9}$	$1.53\cdot10^{-9}$	$2.54\cdot10^{-9}$	$1.22\cdot10^{-9}$	$1.65\cdot10^{-9}$	$3.04\cdot10^{-9}$	$9.89\cdot10^{-9}$	$1.78\cdot10^{-9}$	$3.29\cdot10^{-9}$	$1.67\cdot10^{-9}$
PM	kg PM2.5eq	$2.37\cdot10^{-6}$	$1.65\cdot10^{-6}$	$3.87\cdot10^{-6}$	$2.31\cdot10^{-6}$	$3.50\cdot10^{-6}$	$1.74\cdot10^{-6}$	$2.48\cdot10^{-6}$	$4.06\cdot10^{-6}$	$1.75\cdot10^{-6}$	$2.58\cdot10^{-6}$	$4.51\cdot10^{-6}$	$7.86\cdot10^{-6}$
POF	kg NMVOC eq	$7.96\cdot10^{-5}$	$3.93\cdot10^{-5}$	$1.08\cdot10^{-4}$	$7.41\cdot10^{-5}$	$1.01\cdot10^{-4}$	$4.12\cdot10^{-5}$	$8.24\cdot10^{-5}$	$1.12\cdot10^{-4}$	$4.80\cdot10^{-5}$	$6.46\cdot10^{-5}$	$1.22\cdot10^{-4}$	$1.30\cdot10^{-4}$
AC	molc H+ eq	$6.55\cdot10^{-5}$	$3.42\cdot10^{-5}$	$9.18\cdot10^{-5}$	$6.13\cdot10^{-5}$	$8.48\cdot10^{-5}$	$3.58\cdot10^{-5}$	$6.79\cdot10^{-5}$	$9.55\cdot10^{-5}$	$4.02\cdot10^{-5}$	$5.55\cdot10^{-5}$	$1.04\cdot10^{-4}$	$1.04\cdot10^{-4}$
TE	molc N eq	$2.79\cdot10^{-4}$	$1.22\cdot10^{-4}$	$3.53\cdot10^{-4}$	$2.57\cdot10^{-4}$	$3.34\cdot10^{-4}$	$1.27\cdot10^{-4}$	$2.88\cdot10^{-4}$	$3.65\cdot10^{-4}$	$1.62\cdot10^{-4}$	$2.06\cdot10^{-4}$	$3.94\cdot10^{-4}$	$4.82\cdot10^{-4}$
FE	kg P eq	$7.57\cdot10^{-8}$	$6.86\cdot10^{-8}$	$1.06\cdot10^{-7}$	$7.97\cdot10^{-8}$	$9.89\cdot10^{-8}$	$7.37\cdot10^{-8}$	$8.11\cdot10^{-8}$	$1.13\cdot10^{-7}$	$8.96\cdot10^{-8}$	$1.00\cdot10^{-7}$	$1.40\cdot10^{-7}$	$1.31\cdot10^{-7}$
ME	kg N eq	$2.54\cdot10^{-5}$	$1.11\cdot10^{-5}$	$3.21\cdot10^{-5}$	$2.33\cdot10^{-5}$	$3.03\cdot10^{-5}$	$1.15\cdot10^{-5}$	$2.62\cdot10^{-5}$	$3.31\cdot10^{-5}$	$1.47\cdot10^{-5}$	$1.87\cdot10^{-5}$	$3.57\cdot10^{-5}$	$4.39\cdot10^{-5}$
MFRD	kg Sb eq	$1.29\cdot10^{-7}$	$1.01\cdot10^{-7}$	$2.54\cdot10^{-7}$	$1.21\cdot10^{-7}$	$1.98\cdot10^{-7}$	$1.08\cdot10^{-7}$	$1.37\cdot10^{-7}$	$2.74\cdot10^{-7}$	$1.19\cdot10^{-7}$	$1.57\cdot10^{-7}$	$3.29\cdot10^{-7}$	$2.63\cdot10^{-7}$

Notes: 3M_SA: 3 mouldboard plough, sandy soil; 3M_MT: 3 mouldboard plough, medium texture; 3M_CL: 3 mouldboard plough, clay soil; 3S_MT: 3 slatted plough, medium texture; 3S_CL: 3 slatted plough, clay soil; 3M_SA+: 3 mouldboard plough, sandy soil, different field shape; 3M_MT+: 3 mouldboard plough, medium texture, different field shape; 3M_CL+: 3 mouldboard plough, clay soil, different field shape; 5M_SA: 5 mouldboard plough, sandy soil; 5M_MT: 5 mouldboard plough, medium texture; 5M_CL: 5 mouldboard plough, clay soil; ECO: Ecoinvent ploughing process.

Table 7
Hotspot identification (process emissions, tractor, implement, fuel plus lubricant and shed productions) for the FU for all cases and all impacts categories.

Impact categories	Processes	3M_SA	3M_MT	3M_CL	3S_MT	3S_CL	5M_SA	5M_MT	5M_CL	3M_SA+	3M_MT+	3M_CL+	ECO
CC	Process emissions	73.36%	75.29%	77.35%	74.79%	77.07%	68.37%	74.15%	76.28%	73.25%	75.21%	77.23%	69.56%
	Tractor production	3.98%	3.90%	4.49%	3.65%	3.86%	5.25%	4.11%	5.10%	4.04%	3.96%	4.61%	7.72%
	Plough production	7.04%	5.27%	2.89%	6.10%	3.73%	11.44%	6.59%	3.63%	7.18%	5.40%	2.96%	8.24%
	Diesel fuel and lubricant prod.	13.77%	14.16%	14.51%	14.05%	14.48%	12.85%	13.94%	14.32%	13.78%	14.12%	14.48%	13.22%
	Shed	1.85%	1.38%	0.76%	1.41%	0.86%	2.09%	1.21%	0.66%	1.75%	1.31%	0.72%	1.27%
OD	Process emissions	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
	Tractor production	3.29%	3.17%	3.60%	2.98%	3.10%	4.46%	3.37%	4.11%	3.33%	3.23%	3.70%	6.39%
	Plough production	3.41%	2.51%	1.36%	2.92%	1.76%	5.70%	3.17%	1.71%	3.47%	2.58%	1.40%	4.00%
	Diesel fuel and lubricant prod.	92.48%	93.71%	94.71%	93.47%	94.76%	88.87%	92.93%	93.88%	92.42%	93.62%	94.59%	89.04%
	Shed	0.82%	0.61%	0.33%	0.62%	0.37%	0.96%	0.53%	0.29%	0.78%	0.58%	0.31%	0.57%
PM	Process emissions	20.41%	30.71%	28.39%	29.10%	29.41%	23.37%	24.31%	27.02%	20.21%	30.22%	27.87%	72.96%
	Tractor production	10.09%	9.18%	11.77%	8.63%	9.85%	11.06%	10.18%	13.14%	10.22%	9.36%	12.12%	5.95%
	Plough production	18.86%	13.10%	8.01%	15.24%	10.07%	25.45%	17.23%	9.87%	19.21%	13.48%	8.24%	6.71%
	Diesel fuel and lubricant prod.	45.50%	43.44%	49.64%	43.37%	48.25%	35.29%	45.00%	48.09%	45.50%	43.53%	49.69%	13.30%
	Shed	5.14%	3.57%	2.18%	3.66%	2.42%	4.84%	3.28%	1.88%	4.85%	3.41%	2.08%	1.08%
POF	Process emissions	65.71%	78.33%	72.32%	76.95%	73.70%	72.31%	68.60%	71.13%	65.24%	77.92%	71.85%	83.42%
	Tractor production	3.95%	2.56%	3.95%	2.52%	3.20%	3.77%	3.80%	4.55%	4.05%	2.64%	4.11%	3.37%
	Plough production	6.38%	3.15%	2.32%	3.84%	2.83%	7.50%	5.57%	2.96%	6.58%	3.29%	2.41%	3.29%
	Diesel fuel and lubricant prod.	22.25%	15.11%	20.78%	15.79%	19.59%	15.02%	21.00%	20.81%	22.51%	15.34%	21.03%	9.40%
	Shed	1.71%	0.84%	0.62%	0.91%	0.67%	1.40%	1.04%	0.55%	1.63%	0.81%	0.60%	0.52%
AC	Process emissions	55.27%	69.89%	62.32%	68.23%	64.00%	63.35%	58.41%	61.05%	54.75%	69.39%	61.77%	76.87%
	Tractor production	4.54%	3.10%	4.63%	3.03%	3.79%	4.48%	4.41%	5.32%	4.64%	3.19%	4.81%	4.20%
	Plough production	7.22%	3.76%	2.69%	4.56%	3.30%	8.79%	6.36%	3.41%	7.42%	3.91%	2.79%	4.04%
	Diesel fuel and lubricant prod.	31.04%	22.24%	29.65%	23.11%	28.14%	21.73%	29.62%	29.58%	31.34%	22.54%	29.94%	14.25%
	Shed	1.93%	1.01%	0.72%	1.08%	0.78%	1.64%	1.19%	0.64%	1.84%	0.97%	0.69%	0.64%
TE	Process emissions	77.24%	87.32%	82.79%	86.34%	83.85%	83.26%	79.88%	82.00%	76.85%	87.05%	82.44%	91.09%
	Tractor production	2.07%	1.18%	1.96%	1.18%	1.57%	1.82%	1.94%	2.28%	2.13%	1.23%	2.05%	1.47%
	Plough production	3.96%	1.73%	1.37%	2.14%	1.64%	4.29%	3.37%	1.76%	4.11%	1.81%	1.43%	1.71%
	Diesel fuel and lubricant prod.	14.94%	8.98%	13.25%	9.49%	12.29%	9.28%	13.74%	13.40%	15.19%	9.15%	13.47%	5.28%
	Shed	1.79%	0.78%	0.62%	0.85%	0.66%	1.35%	1.06%	0.56%	1.72%	0.76%	0.60%	0.45%
FE	Process emissions	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
	Tractor production	23.57%	27.98%	42.09%	24.36%	33.92%	21.01%	25.49%	41.22%	23.57%	27.94%	42.33%	34.68%
	Plough production	64.93%	58.86%	42.23%	63.40%	51.10%	71.27%	63.61%	45.66%	65.27%	59.30%	42.44%	57.65%
	Diesel fuel and lubricant prod.	8.05%	10.03%	13.44%	9.27%	12.58%	5.08%	8.54%	11.43%	7.94%	9.84%	13.15%	5.87%
	Shed	3.45%	3.13%	2.24%	2.97%	2.39%	2.64%	2.36%	1.69%	3.21%	2.92%	2.09%	1.80%
ME	Process emissions	77.50%	87.49%	82.95%	86.52%	84.01%	83.57%	80.12%	82.18%	77.12%	87.22%	82.61%	91.27%
	Tractor production	1.96%	1.12%	1.86%	1.12%	1.48%	1.72%	1.84%	2.17%	2.02%	1.16%	1.94%	1.39%
	Plough production	3.79%	1.65%	1.31%	2.03%	1.56%	4.09%	3.22%	1.68%	3.92%	1.73%	1.36%	1.62%
	Diesel fuel and lubricant prod.	15.00%	8.98%	13.28%	9.50%	12.31%	9.31%	13.79%	13.43%	15.26%	9.15%	13.50%	5.27%
	Shed	1.75%	0.76%	0.60%	0.83%	0.64%	1.32%	1.04%	0.54%	1.68%	0.74%	0.58%	0.44%
MFRD	Process emissions	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
	Tractor production	83.05%	85.35%	90.43%	83.30%	87.69%	82.17%	84.39%	90.67%	83.12%	85.41%	90.58%	89.80%
	Plough production	11.15%	8.75%	4.42%	10.56%	6.44%	13.58%	10.26%	4.89%	11.22%	8.83%	4.43%	7.27%
	Diesel fuel and lubricant prod.	4.58%	4.94%	4.66%	5.12%	5.25%	3.21%	4.56%	4.06%	4.52%	4.86%	4.54%	2.45%
	Shed	1.22%	0.96%	0.48%	1.02%	0.62%	1.04%	0.79%	0.37%	1.14%	0.90%	0.45%	0.47%

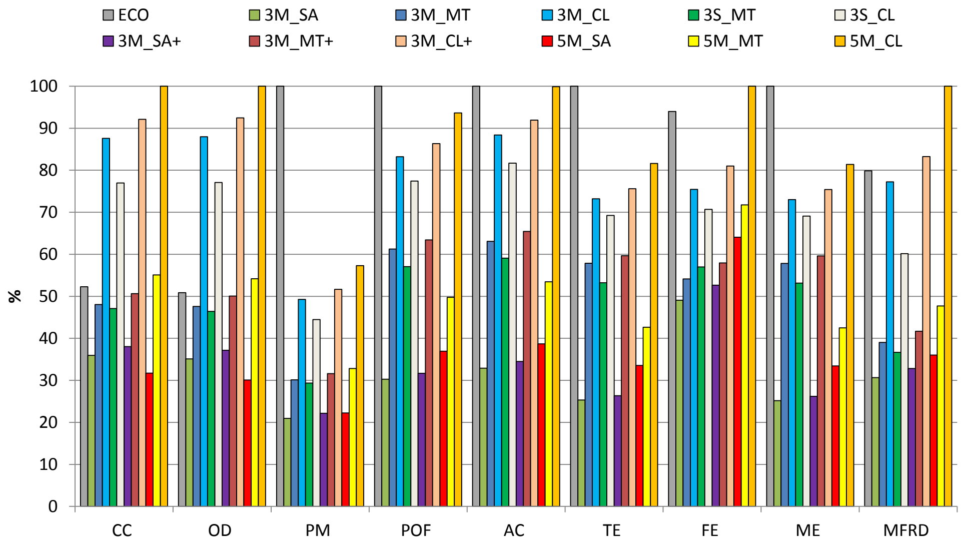

Fig. 4. Relative comparison of the analysed primary soil tillage operations for the evaluated impact categories.

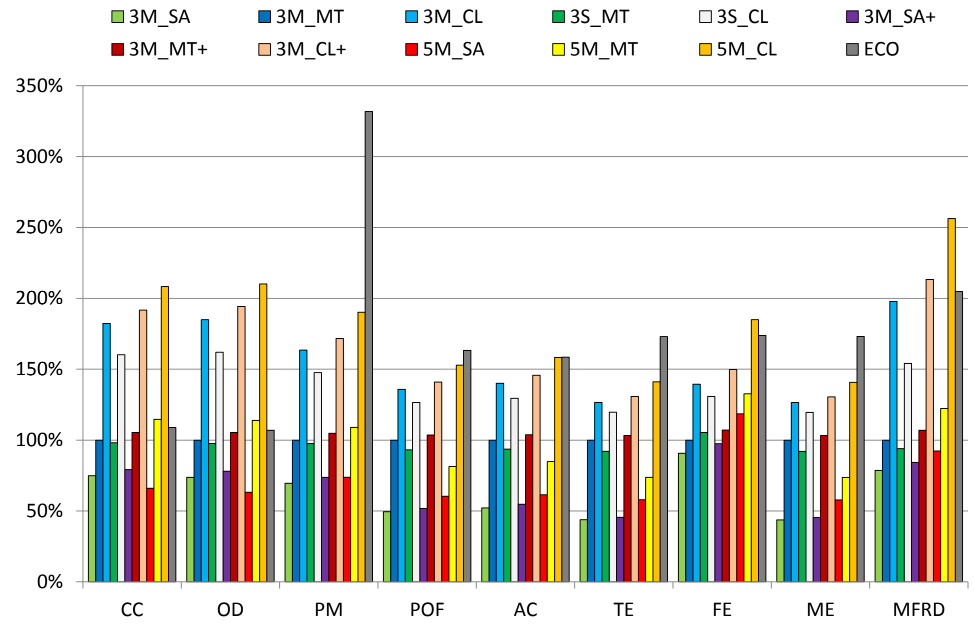

Fig. 5. Environmental impacts comparison of the primary soil tillage operations. All cases are referred to the 3 ploughshares mouldboard plough on medium texture soil (3M_MT) as reference (100%).

highest impacts were from Ecoinvent process (ECO), showing an increase of +63.2%, +58.5% and +72.8%, respectively, compared to 3M_MT.

(v) ME also showed the highest impact reduction for the 3 ploughshares mouldboard plough on sandy soil (3M_SA), which was −56.4% of 3M_MT, and the highest impact increase for Ecoinvent process (ECO), which was +73.0% of 3M_MT. In all these 4 impact categories, process emissions (emissions to air and soil, materials consumed, tyre abrasion and fuel and lubricant consumption) played a prominent role.

(vi) In FE, the 3 ploughshares mouldboard plough on sandy soil (3M_SA) had an impact reduction of −9.3% compared to reference 3M_MT. On the opposite, the impact increased the most for the 5 ploughshares mouldboard plough on clay soil (5M_CL), +84.8% when compared to 3M_MT. On this impact category, tractor and plough production (and, consequently, amounts of materials consumed) had a prominent role. In more details, masses of materials consumed were assessed in each case completed with ENVIAM, therefore, smaller or bigger machines masses were considered. Both implement and tractor were characterised by a lower amount of materials deteriorated when the 3 ploughshares plough was considered instead of the 5 ploughshares. If the same number of ploughshares was considered, this assertion was even more effective for a slatted plough. On the opposite, a mouldboard plough with two more ploughshares, as in 5M_CL, was characterised by heavier mass and required a more powerful tractor (bigger masses consumed).

(vii) Finally, MFRD highest impact reduction was due to the 3 ploughshares mouldboard plough on sandy soil (3M_SA), with a reduction of −21.5% compared to the reference 3M_MT. The highest impact increase was due to the 5 ploughshares mouldboard plough on clay soil (5M_CL), with an increase of +156.2% when compared to reference. The reason of this huge discrepancy was first the tractor production and secondly the implement production. They entailed materials production and energy used for production that increased at the increase of masses involved.

Even if the reference case 3M_MT (3 ploughshares mouldboard plough on medium texture) had similar assumptions to Ecoinvent ploughing process (ECO), the differences between the results of the two impact assessments were marked, the second being even one of the two most impacting analysed cases. ECO was the process that, together with 5 ploughshares mouldboard plough on clay soils (5M_CL), had the highest impacts on most impact categories. On the opposite3M_MT was the most impacting process for no impact category.

3.1. Impact of soil texture

Considering the same groups of implements (either 3 ploughshares mouldboard or 3 ploughshares slatted or 5 ploughshares mouldboard ploughs), working in conditions with diverse soil textures, differences in their environmental impacts were not negligible. In all cases, sandy soils had environmental impacts that were between 9% and 56% less than medium texture ones, while ploughing in clay soils showed impacts being 41%–156% higher than the same operation carried out on a medium texture soil. These effects were mainly related to fuel consumption and tractor engine power.

The comparison between Ecoinvent and ENVIAM showed that differences could already be highlighted when the reference case 3M_MT was assessed. The most evident differences in impacts between these two were due to PM and MFRD, both resulting from fuel consumption and tractor production. For example, fuel consumption in ENVIAM ploughing processes ranged between 15.4 $kg \cdot ha^{-1}$ for the least impacting ploughing operation and 54.1 $kg \cdot ha^{-1}$ for the most impacting. 3M_MT, however, showed a difference with Ecoinvent process lower than 2%. Also POF, AC, ME, TE and FE showed much higher impacts in Ecoinvent process when compared to 3M_MT, due to fuel and lubricant consumption and emissions to atmosphere, but also to machinery lifespan and materials masses consumed (differences of environmental impacts between the least and most impacting options equal to 77.0%).

3.2. Impact of a different field shape ratio

With regard to the cases of the 3 ploughshares mouldboard plough on sandy soil (3M_SA+), on medium texture soil (3M_MT+) and on clay soil (3M_CL+) in which the field shape changed from a rectangular (50 m × 200 m) to a square shape (100 m × 100 m), the working time increased (from 1.61 $h \cdot ha^{-1}$ to 1.71 $h \cdot ha^{-1}$). The reason behind this was that a higher amount of turns at headlands was necessary, which increased the duration of the operation. Turns were performed in a condition of low engine load (40%), which affected the outputs (fuel and exhaust gases emissions). In addition, the varied shape ratio influenced all mechanical inputs directly affected by working time (e.g., lubricant, tractor and implement consumptions). In this condition, environmental impacts increased in all categories, if compared with the related cases with rectangular field shape ratio (3M_SA, 3M_MT, 3M_CL). For all impact categories, the impact increase ranged between 3.1% (POF, TA, TE and ME) and 8.0% (CC, OD, PM, FE and MFRD) of the similar option with no varied field shape ratio. In particular, TE and ME were the categories less influenced by field shape ratio change (+3.1%, +4.1%, +3.3%, for 3M_SA+, 3M_MT+ and 3M_CL+, respectively) because the increase in fuel consumption and in emissions to air and soil were not significantly relevant. MFRD showed the most evident effect (+6.9%, +7.2% and +7.9% for 3M_SA+, 3M_MT+ and 3M_CL+, respectively), because of tractor and plough consumption.

3.3. Impact of the choice of a wider implement

Three studied cases concerned 5 ploughshares mouldboard ploughs (on sandy soil: 5M_SA, on medium texture soil: 5M_MT and on clay soil: 5M_CL). They were compared with the 3 ploughshares mouldboard ploughs (3M_SA, 3M_MT, 3M_CL, on sandy, medium texture and clay soils, respectively), highlighting the following differences. CC and OD showed better environmental results for 5 ploughshares plough on sandy soil (5M_SA) if compared with 3 ploughshares mouldboard plough on sandy soil (3M_SA), explained by the fact that fuel consumption was 7.7% lower in 5 ploughshares plough than in 3 ploughshares plough. On the opposite, all other impact categories had better results with the 3 ploughshares mouldboard plough on sandy soil (3M_SA), since less materials were entailed for production and consumption. The case of the 5 ploughshares plough with clay soil (5M_CL) was one of the most impacting, being characterised by the most difficult working conditions (e.g., highest soil resistance to overcome, highest engine power needed, highest masses of tractor and plough involved). The most evident differences between 5 and 3 ploughshares mouldboard ploughs were attributed to MFRD (range between +17.6% and +29.5%) and FE (range between +30.6% and +32.6%), due the tractor and plough production. This meant that, for these cases, the reduction in total working time that was consequent to the wider implement (wider working width and less turns at headlands for 5 ploughshares ploughs), had no significant effect. With regard to the sandy soil condition, POF, AC, TE and ME showed environmental effects lower for the 5 ploughshares mouldboard plough (5M_SA) than for the 3 ploughshares mouldboard plough on sandy soil (3M_SA), because working time, fuel and lubricant consumed and emissions to atmosphere and soil affected the most the environmental impacts.

4. Discussion

In the case studies of different primary soil tillage operations, the evaluation of the environmental impacts highlighted interesting results. A comparison was carried out between the inventory filled in with Ecoinvent ploughing process and the one filled in with ENVIAM. In ENVIAM, local conditions (soil texture, field shape ratio and size) and working time were considered in order to evaluate their influence on the environmental impact. Both soil texture and logistic issues (field shape and size) affected ploughing. This operation shows large discrepancies with Ecoinvent database because the different local conditions (soil and logistic) cannot be varied. Differently, in the proposed case study, the comparison among ploughing performed on fields with different soil texture, shape ratio and size (e.g., impacts in 3M_SA+, 3M_MT+ and 3M_CL+ compared with 3M_SA, 3M_MT, 3M_CL) showed the relevance of these parameters on the environmental impact. On the opposite, other operations such as application of products (e.g., fertilisers, pesticides), harvesting and transport are influenced only by logistic conditions (field shape ratio and size). Consequently, for these operations, the influence of logistic conditions is expected to be similar to the one highlighted for ploughing performed in fields with equal soil conditions but different field shape ratio and size.

In the comparison between Ecoinvent ploughing and the one assessed by ENVIAM in similar conditions (3 ploughshares mouldboard plough and medium texture soil - 3M_MT) the discrepancies were low for CC and OD (<9%) but high for the other impact categories, especially for PM and MFRD (2–3 times higher). With regard to the comparison carried out within ENVIAM, many differences were highlighted only considering different soil textures. The consequences of the different soil texture and working time were the different engine power requests, fuel consumptions as well as materials and lubricant consumptions and exhaust gases emissions. In particular, the increase in soil resistance forces when the soil changed from sandy to medium texture and clay, respectively, caused increases of environmental effects. With regard to fuel consumption and exhaust gases emissions, the analysis of power and load done in ENVIAM was used to reach higher accurateness in the LCI. In fact, the main lack of Ecoinvent database is the absence of inventories performable in multiple working conditions. Contrarily to industrial standardized processes, for agricultural ones, these lacks are influential because processes are subjected to natural issues difficult to consider in a database. When the local conditions were assessed, the environmental impact discrepancy resulting from Ecoinvent ploughing process showed inadequate outcomes, the average being too high for sandy soils and too low for clay soils.

5. Conclusions

The attention on the environmental issues linked to agricultural machinery has increased enormously. Databases and tools are available for research and commercial users and are very useful when primary data are not available. However, in particular in the agricultural sector, the field production processes are complex. For this reason, databases can offer misleading information when not reflecting the actual study conditions. Consequently, when local conditions differ from secondary data obtained from common database averages (Ecoinvent), it is important for LCA practitioners to use either primary data or secondary ones estimated considering local conditions. Inventory data in LCA studies are essential to be reliable because environmental impacts are quantified for a defined system that is characterised by distinct inventory information. When data do not represent the system, environmental impacts do neither, and other sources should be used. In general terms, primary data are always the most preferable; nevertheless, when the study is focused on agricultural systems, and secondary data are used, they must be based on local conditions. The use of secondary data from database for LCA study should occur only when the agricultural systems represent only a part of the assessed production systems. In order to avoid unrepresentative environmental impacts, what emerges from this study, is the recommendation of using primary or secondary data evaluated considering local conditions.

In this study, the comparison between the environmental assessment made fulfilling the LCI with ENVIAM and with Ecoinvent database v3 highlighted that the average values present in Ecoinvent ploughing process are applicable only in similar conditions to those averages. On the opposite, when soil and implement have diverse features, the differences are not negligible. Environmental impacts are deeply affected by this choice, both in positive terms (the operative choice entails a benefit if compared to the one deriving from the use of Ecoinvent ploughing process), as well as in negative terms (when the operative choice entails a steep increase in the environmental impacts, if compared with the one deriving from the inventory with Ecoinvent ploughing process). Therefore, the use of tools adaptable to local contexts, gives the opportunity to obtain an improvement in the environmental assessments and sustainability evaluations.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.02.011.

References

Bacenetti, J., Fusi, A., Guidetti, R., Fiala, M., 2013a. Life Cycle Assessment of maize cultivation for biogas production. J. Agric. Eng. XLIV, 579–582.

Bacenetti, J., Negri, M., Fiala, M., Gonzalez-Garcia, S., 2013b. Anaerobic digestion of different feedstocks: impact on energetic and environmental balances of biogas process. Sci. Total Environ. 463–464, 541–551.

Bacenetti, J., Fusi, A., Fiala, M., 2014a. Impatto ambientale delle operazioni agricole di campo: effetto della scelta della macchina operatrice e delle condizioni pedologiche. In: VIII Convegno Rete Italiana LCA, pp. 19–20. June 2014, Florence.

Bacenetti, J., Fusi, A., Negri, M., Guidetti, R., Fiala, M., 2014b. Environmental assessment of two different crop systems in terms of biomethane potential production. Sci. Total Environ. 466–467, 1066–1077.

Bacenetti, J., Fusi, A., Negri, M., Fiala, M., 2015a. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Bacenetti, J., Lovarelli, D., Ingrao, C., Tricase, C., Negri, M., Fiala, M., 2015b. Assessment of the influence of energy density and feedstock transport distance on the environmental performance of methane from maize silages. Bioresour. Technol. 193, 256–265.

Bengoa, X., Rossi, V., Humbert, S., Nemecek, T., Lansche, J., Mouron, P., 2014. Methodological guidelines for the life cycle inventory of agricultural products. World Food LCA Database 1–79.

Blengini, G.A., Busto, M., 2009. The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). J. Environ. Manag. 90 (3), 1512–1522.

Bodria, L., Pellizzi, G., Piccarolo, P., 2006. Il trattore e le macchine operatrici. Edagricole.

Bolli, P., Scotton, M., 1987. Lineamenti di tecnica della meccanizzazione agricola. Edagricole, Bologna.

Brentrup, F., Küsters, J., Kuhlmann, H., Lammel, J., 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology. Eur. J. Agron. 20 (3), 247–264.

CEC (Commission of the European Communities), 2003. 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.

Chiaramonti, D., Recchia, L., 2010. Is life cycle assessment (LCA) a suitable method for quantitative CO_2 saving estimations? The impact of field input on the LCA results for a pure vegetable oil chain. Biomass Bioenerg. 34 (5), 787–797.

Dace, E., Muizniece, I., Blumberga, A., Kaczala, F., 2015. Searching for solutions to mititgate greenhouse gas emissions by agricultural policy decisions – application of system dynamics modeling for the case of Latvia. Sci. Total Environ. 527–528, 80–90.

Dyer, J.A., Desjardins, R.L., 2003. Simulated farm fieldwork, energy consumption and related greenhouse gas emissions in Canada. Biosyst. Eng. 85 (4), 503–513.

Ecoinvent database. http://www.ecoinvent.org/database/. (accessed 27.06.15.).

European Directive 97/68/EC, 1998. On emissions from non-road mobile machinery. Off. J. Eur. Communities L 59, 1.

Fallahpour, F., Aminghafouri, A., Ghalegolab Behbahani, A., Bannayan, M., 2012. The environmental impact assessment of wheat and barley production by using life cycle assessment (LCA) methodology. Environment. Dev. Sustain. 14 (6), 979–992.

Falloon, P., Betts, R., 2010. Climate impacts on European agriculture and water management in the context of adaptation and mitigation-The importance of an integrated approach. Sci. Total Environ. 408 (23), 5667–5687.

Fedele, A., Mazzi, A., Niero, M., Zuliani, F., Scipioni, A., 2014. Can the Life Cycle Assessment methodology be adopted to support a single farm on its environmental impacts forecast evaluation between conventional and organic production? An Italian case study. J. Clean. Prod. 69, 49–59.

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.

Goedkoop, M., De Oele, M., Leijting, J., Ponsioen, T., Meijer, E., 2013. Simapro Introduction to LCA with SimaPro 7 (Pré Consultants).

Hansson, P.A., Lindgren, M., Norén, O., 2001. A comparison between different methods of calculating average engine emissions for agricultural tractors. J. Agric. Eng. Res. 80 (1), 37–43.

Hokazono, S., Hayashi, K., 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. J. Clean. Prod. 28, 101–112.

Ingrao, C., Matarazzo, A., Tricase, C., Clasadonte, M.T., Huisingh, D., 2015. Life Cycle Assessment for highlighting environmental hotspots in Sicilian peach production systems. J. Clean. Prod. 92, 109–120.

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.

ISO 14040 series, 2006. Environmental Management – Life Cycle Assessment – Principles and Framework.

ISO 8178-4 cycle C1, International Standard for Exhaust Emission Measurement.

ISTAT, 2015. Available at. http://dati-censimentoagricoltura.istat.it/?lang=it. accessed 01.09.15.

Jannick, H.S., Weidema, B.P., Muñoz, I., Dalgaard, R., Merciai, S., de Saxcé, M., 2010. 2.0 LCA Consultants. www.lca-net.com (accessed 27.06.15.).

Janulevičius, A., Juostas, A., Pupinis, G., 2013a. Engine performance during tractor operational period. Energ Convers. Manag. 68, 11–19.

Janulevičius, A., Juostas, A., Pupinis, G., 2013b. Tractor's engine performance and emission characteristics in the process of ploughing. Energ Convers. Manag. 75, 498–508.

Jensen, A.A., Hoffman, L., Møller, B.T., Schmidt, A., 1997. Life Cycle Assessment a Guide to Approaches, Experiences and Information Sources. European Environment Agency.

Kerkhof, A., 2012. Life Cycle-based Sustainability – Standards and Guidelines (Pré Consultants).

Keyes, S., Tyedmers, P., Beazley, K., 2015. Evaluating the environmental impacts of conventional and organic apple production in Nova Scotia, Canada, through life cycle assessment. J. Clean. Prod. 104, 40–51.

Kim, Y.-J., Chung, S.-O., Choi, C.-H., 2013. Effects of gear selection of an agricultural tractor on transmission and PTO load during rotary tillage. Soil Till Res. 134, 90–96.

Lacour, S., Burgun, C., Perilhon, C., Descombes, G., Doyen, V., 2014. A model to assess tractor operational efficiency from bench test data. J. Terramechanics 54, 1–18.

Lazzari, M., Mazzetto, F., 2005. Prontuario di meccanica agraria e meccanizzazione. REDA.

Lindgren, M., Larsson, G., Hansson, P.A., 2010. Evaluation of factors influencing emissions from tractors and construction equipment during realistic work operations using diesel and bio-fuels as substitute. Biosyst. Eng. 107, 123–130.

Lovarelli, D., Bacenetti, J., Fiala, M., 2015. A new tool for life cycle inventories of agricultural machinery operations. J. Agric. Eng. (in press).

Meul, M., Middelaar, C.E., de Boer, I.J.M., Van Passel, S., Fremaut, D., Haesaert, G., 2014. Potential of life cycle assessment to support environmental decision making at commercial dairy farms. Agric. Syst. 131, 105–115.

Nemecek, T., Kägi, T., 2007. Life cycle inventory of agricultural production systems. Ecoinvent report 15. Ecoinvent report, 2007. Swiss Centre for Life Cycle Inventories, Dübendorf.

Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., Schaller, B., Chervet, A., 2011. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agric. Syst. 104 (3), 233–245.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard, R., Halberg, N., 2003. LCA Food Database (accessed 27.06.15.).

Niero, M., Ingvordsen, C.H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M.F., Hauschild, M.Z., Jørgensen, R.B., 2015. Eco-efficient production of spring barley in a changed climate: a Life Cycle Assessment including primary data from future climate scenarios. Agric. Syst. 136, 46–60.

Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K., 2015. Life Cycle Assessment in the Agri-Food Sector. Springer, pp. 1–390.

Noya, I., González-García, S., Bacenetti, J., Arroja, L., Moreira, M.T., 2015. Comparative life cycle assessment of three representative feed cereals production in the Po Valley (Italy). J. Clean. Prod. 2015, 1–16.

Organization for Economic Co-operation and Development Reports Code 2. 2010 – 2012.

Ossés de Eicker, M., Hischier, R., Kulay, L.A., Lehmann, M., Zah, R., Hurni, H., 2010. The applicability of non-local LCI data for LCA. Environ. Impact Assess. Rev. 30 (3), 192–199.

Schäffeler, U., Keller, M., 2008. Non-road Fuel Consumption and Pollutant Emissions. Study for the period from 1980 to 2020, p. 171 (Bern).

Tendall, D.M., Gaillard, G., 2015. Environmental consequences of adaptation to climate change in Swiss agriculture: an analysis at farm level. Agric. Syst. 132, 40–51.

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. ILCD Handbook-towards More Sustainable Production and Consumption for a Resource Efficient Europe. JRC Reference Report.

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Seedbed preparation for arable crops: Environmental impact of alternative mechanical solutions

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ABSTRACT

This study quantifies, using the Life Cycle Assessment (LCA) method, the environmental impact of sequences of operations for seedbed preparation for arable crops under different soil textures and soil refinement intensities. Comparing the environmental loads of alternative sequences of operations permitted to detect the most environmentally sustainable sequence in each working condition.

To this purpose, 13 alternative sequences of field operations for seedbed preparation were analysed, considering primary and secondary soil tillage and minimum tillage. The study was carried out considering a cradle to farm gate perspective and selecting as functional unit "1 ha tilled with an appropriate soil refinement for sowing and seed germination". Three 60-ha farms in Northern Italy were considered. Inventory data (e.g., farm and field information, machinery fleets, fuel, lubricant and materials consumptions and exhaust gases emissions) were calculated with the model ENVIAM (ENVironmental Inventory of Agricultural Machinery operations).

The impact assessment was completed with the ReCiPe characterisation method. Results showed that seedbed preparation completed with two implements (one for primary and one for secondary soil tillage) instead of three (e.g., one for primary and two for secondary soil tillage) results in a better environmental performance on impact categories (e.g., eutrophication, ecotoxicity and metals depletion) affected by the manufacturing phases and by the consumption of materials along machinery life span. The impact categories affected by fuel consumption and exhaust gases emissions showed the best results with low energy-consumptive operations (e.g., slatted plough, no-Power Take Off harrow and minimum tillage). Coarse textured soils and soils lowly refined (i.e. unrefined soil particles adapt for crops characterized by seeds with a size close to that of winter crops or by high seed density) showed low burdens on all impact categories, whereas fine textured and highly refined soils (i.e. small soil particles adapt for crops with small seeds or by low seed density) were responsible for the highest impacts. This is primarily due to the larger number of harrowing repetitions and of energy and fossil fuel consumption.

The results can be up-scaled to arable crop production systems with similar pedo-climatic and operative features, such as other Mediterranean countries. Farmers associations, stakeholders and politicians could promote policies and define incentives that encourage producers to adapt to more environmentally sustainable crop productions.

1. Introduction

In agriculture, mechanisation represents an essential activity for the cultivation of crops with a marketable value for food and feed purposes. Among mechanical field operations, seedbed preparation is fundamental for achieving an optimal soil condition for seed germination.

Several tillage machines and different sequences of operations for seedbed preparation can be identified (Bacenetti et al., 2015a; Castanheira et al., 2010; Kouwenhoven et al., 2002; Panettieri et al., 2013; Vakali et al., 2011). The reason for this wide range of possibilities (namely primary and secondary soil tillage) is linked to: (i) the specific features characterising the production context (Van Linden and Herman, 2014), (ii) local pedo-climatic variables (e.g., soil texture, soil moisture, field shape and slope, temperature and rainfall) (Lovarelli et al., 2017), (iii) mechanical variables of tractors (e.g., engine power, engine load, working speed, displacement system) (Molari et al., 2012; Perozzi et al., 2016) and implements (e.g., working width, working depth) (Lovarelli and Bacenetti, 2017) and (iv) operating variables (e.g., machinery fleet, working promptness, number of workers, farm-field distance) (Pitla et al., 2016; Šarauskis et al., 2014).

From an environmental perspective, every production system is responsible for an environmental impact. With environmental impact is

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meant the possible adverse effect caused by the subtraction of inputs and release of substances in the environment (i.e. in air, soil and water). The existence of different alternatives in terms of tillage machines and of variables that affect the production context (i.e. pedo-climatic, mechanical and operating variables), determines that different environmental impacts arise. Accordingly, a tractor and an implement, or more in general a sequence of field operations (e.g., primary plus secondary soil tillage), can be adequate for a context, but not for another even spatially close to the first. If more sequences are feasible in the same context, their environmental impact can differ due to this local variability as well as to the availability of machinery options among which to choose (Barthelemy et al., 1992).

Potential adverse effects are quantified with the Life Cycle Assessment (LCA) method. LCA is a worldwide adopted approach (ISO 14040 series, 2006) that aims at investigating with a holistic approach all inputs and outputs of a production system and at transforming these data (i.e. inventory data) in potential environmental impacts. From the several studies performed with LCA approach on the environmental evaluation of agricultural food and feed productions (Notarnicola et al., 2015) it has emerged that field operations are responsible for a relevant share of the environmental load attributed to agriculture (Dace et al., 2015; Hokazono and Hayashi, 2012; IPCC, 2006). Mechanical field operations (e.g., soil tillage, seeding, fertilisation, harvest), are responsible for the consumption of important masses of fuel, of the emission of engine exhaust gases (carbon dioxide CO_2, nitrogen oxides NO_X, carbon monoxide CO, hydrocarbons HC, etc.) and of the consumption of materials (tires abrasion, materials that must be substituted along the machinery life span and those that must be disposed of at the end of the machinery life span, etc.) (Boone et al., 2016; Bacenetti et al., 2015c; Lee et al., 2016; Schmidt Rivera et al., 2017). Each of these substances is responsible for environmental impacts.

The aim of this study is to calculate, using the LCA approach, the environmental load of mechanical field operations for seedbed preparation of arable crops evaluating the effect that the different pedoclimatic, mechanical and operating conditions can have on the environmental point of view. To do this, different seedbed preparation options (primary + secondary soil tillage) were considered. They were defined by taking into account that arable crops can demand for highly or lowly refined soils and for deeper or shallower primary soil tillage.

Several LCA studies have already been performed to quantify the environmental impact of crop production and, although the system boundary often differed, one consistent finding was that mechanisation of seedbed preparation is responsible for a relevant share of the environmental impact of those production systems (Bacenetti et al., 2015a, 2015c; Niero et al., 2015; Noya et al., 2015). Therefore, to identify the operational sequence that present a more environmentally sustainable behaviour, 13 alternative sequences of operations for seedbed preparation of arable cropping systems composed of primary and secondary soil tillage have been assessed focusing on the Italian productive context.

The results of this study can be helpful for farmers' associations, stakeholders and politicians to promote policies and define incentives for the completion of operations with higher environmental benefits, in view of promoting more sustainable productions in the near future. Similarly, the upscaling to other geographic contexts characterised by similar pedo-climatic and operating features such as other Mediterranean countries can be particularly useful and interesting for policies promotion.

2. Materials and methods

The sequences of operations for seedbed preparation selected for this study were analysed with the Life Cycle Assessment (LCA) method (ISO 14040, 2006). LCA is a standardised approach adopted worldwide for quantifying the potential environmental impact of processes for products or services during their whole life cycle using a holistic approach. In LCA, four steps are typically taken:

(i) goal of the study, selection of the functional unit, description of the system and of the system boundary,
(ii) Life Cycle Inventory data collection, aimed to identify and quantify the flow of materials and energy from the studied systems and the environment,
(iii) Life Cycle Impact Assessment, during which inventory data are converted in few numeric indicators of environmental impact thanks to a characterisation method. Several characterisation methods were developed over the years but, for agricultural systems, ReCiPe (Goedkoop et al., 2008) and ILCD (Wolf et al., 2012) are the most applied. Within the characterisation method, for each impact category, different characterisation factors allow the conversion of the inputs (e.g., mass of fuel consumed) and outputs (e.g., emission of pollutant *i* in the atmosphere) in the environmental impact,
(iv) interpretation of the results and identification of the process hotspots.

How these steps were implemented in this study is outlined below.

2.1. Goal and scope

The goal of this study is to quantify the environmental impact of several sequences of field operations for seedbed preparation of arable crops to identify those with a lower environmental burden. With sequence of field operations is meant the set of primary and secondary soil tillage operations to carry out in order to prepare the soil to sowing. Along the manuscript, the term "sequence" will be adopted to identify this set.

The focus is on primary and secondary soil tillage, which represent the most energy-consumptive operations for crop production systems (Gronle et al., 2015; Mileusnić et al., 2010). In this study, field operations were carried out by farmers and not by contractors; therefore, the identified sequences were built upon Italian farm-scale contexts.

2.2. Functional unit and system boundary

During this phase, the Functional Unit (FU) must be defined. The FU describes the function of the system and represents the unit to which all inputs and outputs are referring. Commonly, LCA studies about crop cultivation assess the environmental impact using 1 ha as FU (Nemecek et al., 2015; Solinas et al., 2015) or, alternatively, 1 t. In this study, no specific crop is evaluated; therefore, the selected FU is "1 ha tilled with an appropriate soil refinement for sowing and seed germination".

Additionally, in this phase, also the system boundary must be defined; in the system boundary is stated what is included in the assessment and what is excluded. In this case, it comprises primary (soil overthrow/breaking with no repetition on the same field) and secondary (soil refinement with one or more repetitions on the same field) soil tillage, as illustrated in Fig. 1. Given the recent importance attributed to minimum tillage, sequences of operations characterised by no ploughing were analysed; however, in order to respect the identified FU, interventions on field had to guarantee the appropriate soil refinement and seed germination.

2.3. Description of the system

Scientific literature, experts, farmers' associations and technical journals were queried about the most common sequences of mechanical operations that characterise seedbed preparation of arable crops in Italy. The options disclosed to farmers for field operations were similar to different countries (Bacenetti et al., 2015a; Çarman, 1997; De Vita et al., 2007; Dimanche and Hoogmoed, 2002; Lazzari and Mazzetto,

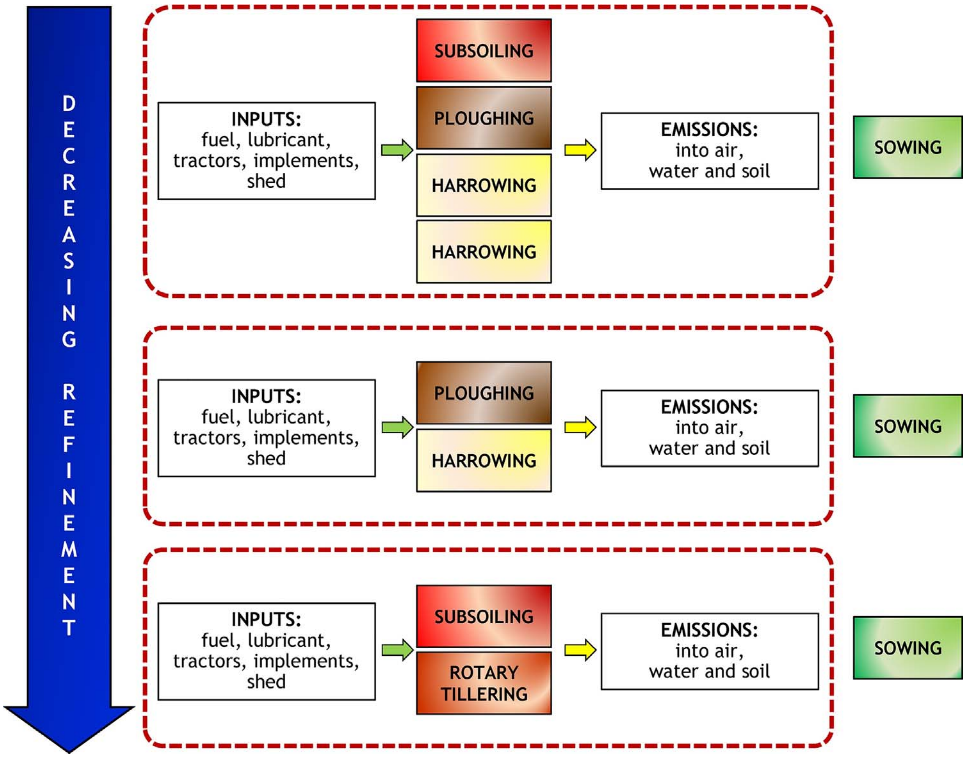

Fig. 1. System boundary with decreasing soil refinement. Inputs and emissions are included, while sowing is excluded.

2005; Marques Da Silva et al., 2004; Noya et al., 2015).

To identify the field operations in this study, the conceptual steps were: (i) identifying the most common crops cultivated in Italy, (ii) completing interviews and queries to collect information about the most common sequences of field operations for seedbed preparation for these identified crops, (iii) grouping the sequences as function of a common ground (hereafter explained), and (iv) quantifying and comparing their environmental impact. As reported in Table 1 the grouping of sequences was made first considering that crops can demand for:

(i) deep ploughing (35 cm, mainly for spring-summer crops) or shallow ploughing (25 cm, mainly for winter cereals),
(ii) high soil refinement intensity (e.g., maize, soybean, sunflower, sorghum, rape and vegetables) typical for spring-summer crops, or low soil refinement intensity (e.g., bread wheat, durum wheat, barley and triticale) for winter crops.

A second distinction was made in terms of soil texture, and three major categories were considered:

(i) coarse textured, characterised by soil particles that break easily by field operations, the latter thus being less energy-consumptive;
(ii) medium textured, characterised by intermediate and generally optimal physical features for crop growth;
(iii) fine textured, where soil particles are cohesive and, therefore, require more energy-consumptive operations (e.g., high tractor engine power and demanding refinement capabilities of the tillage equipment).

Table 1
List of the queried crops with information about the related seedbed cultivation practice (i.e. ploughing depth and refinement intensity). On the right a code has been attributed to each of them.

Crop	Ploughing depth		Refinement		Code
	≈ 25 cm	≈ 35 cm	Low (L)	High (H)	
Maize		x		x	H_{35}
Soybean	x			x	H_{25}
Sunflower		x		x	H_{35}
Sorghum	x			x	H_{25}
Rape	x			x	H_{25}
Bread wheat	x		x		L_{25}
Spring wheat	x		x		L_{25}
Barley	x		x		L_{25}
Triticale	x		x		L_{25}
Vegetables	x			x	H_{25}

A general assumption is that the finer textured the soil is, the higher tractor engine power to carry out both primary and secondary soil tillage and the higher the number of repetitions for secondary soil tillage are necessary.

In all the studied sequences, soil moisture was assumed to favour optimal conditions for tilth, in agreement with recommendations for best working conditions (Ahmadi and Mollazade, 2009; Šarauskis et al., 2014). As regard to tractors and implements availabilities and selections, it was assumed that operations were carried out by farmers equipped with average machinery fleets and that no operation was carried out by contractors. The implements present in each sequence of operations were selected and analysed using the model ENVIAM (ENVironmental Inventory of Agricultural Machinery operations) (Lovarelli et al., 2016), a tool developed to support the filling of inventories for agricultural machinery operations.

Since not every implement was adequate for any soil texture (Table 2), the following operations were evaluated:

- primary soil tillage, which included:

a subsoiler as alternative to ploughing in conservation tillage;
b plough, mainly mouldboard, as it was the most common; slatted plough was used on medium texture and fine textured soils to reduce high traction forces while disc plough was uncommon on fine textured soils;
c rotary tiller was commonly not used on coarse textured soils, since such an energy-consumptive implement is discouraged on these soils. This implement could be used for either primary or secondary soil tillage;

- secondary soil tillage, which included:

a spring tine harrows, fixed teeth harrows and disc harrows. All of them are no power-take-off (PTO) implements combined with rollers and are

Table 2
Implements that have been adopted in the studied system for the soil texture typologies ("X" allowed; "–" not allowed).

Implement	Coarse textured soil	Medium texture soil	Fine textured soil
Subsoiler	X	X	X
Mouldboard plough	X	X	X
Disc plough	X	X	–
Slatted plough	–	X	X
Rotary tiller	–	X	X
Spring tine harrow	X	X	X
Fixed teeth harrow	X	X	X
Disc harrow	–	X	X
Rotary harrow	–	X	X

common on all soil textures. They require, however, a different number of repetitions on the same field according to texture and refinement. From the interviews, it has emerged that the most common implement was the spring tine harrow; fixed teeth harrows only achieved a good result on coarse textured soils when more repetitions were done. Disc harrows were frequently employed when the soil particles after ploughing were still big and needed an intermediate refinement step;

b rotary harrow. It employed PTO and had interesting refinement quality, although it was more energy-consumptive than other no-PTO harrows. Similarly to rotary tiller, its use was uncommon on coarse textured soils, since other implements could obtain a similar result with lower fuel consumption and power requests.

The common seedbed preparation solution entailed one primary soil tillage operation and one secondary soil tillage operation that could be carried out with one or more implements and in one or more repetitions. Alternatively, even if still less practised, minimum tillage solutions have been analysed (De Vita et al., 2007; Alvarez and Steinbach, 2009). Among all the studied alternatives, the sequences that emerged more than once were removed from the analysis to avoid identical repetitions of arrangements.

The studied sequences, showing information about the typology of implement and number of repetitions, are shown in Table 3. For each sequence, a distinction was made among soil texture category and primary and secondary soil tillage grouping (i.e. ploughing depth and refinement intensity). A code is reported: in the sequences, when the same letters are shown it means that the same sequence was carried out.

2.4. Life Cycle Inventory

In each sequence, the tractor power needed to carry out the operation and the machinery inventory data were calculated with the model ENVIAM (Lovarelli et al., 2016, 2017), described in Section 2.4.1.

Inventory data were assessed for three farms located in the Italian Po Valley, each characterised by 60 ha of arable irrigated land devoted to cereal crop cultivation. The farms were selected according to soil texture: one characterised by coarse textured soils (named "F_A", located in the District of Alessandria), the second by medium texture soils (named "F_M", located in the District of Lodi) and the third by fine textured soils (named "F_F", located in the District of Ferrara). The dimension of the farm and the soil texture category affected the farms' machinery fleets, which were composed of three 4WD tractors (on average 5–6 kW/ha) for seedbed preparation with features summarised in Table 4. Tractors' engines belonged to the emissive Stage IIIB (EU Directive 97/68/EC; 2010/26/EU; 2010/22/EU) and exhaust gases emissions were calculated accordingly. Their characteristics (mass, power, life span, specific fuel consumption, etc.) as well as those of implements were available in ENVIAM. The annual working time (h/yr) was assumed considering the average value for tractors working on cereal farms. Seedbed preparation was considered being repeated twice per year, since the three farms were characterised by a double cultivation system that is a common practice in the Po Valley (Bacenetti et al., 2015a; Negri et al., 2014a, 2014b). The annual working time reported in Table 4 was close to the Italian average available in Calcante et al. (2011) for two groups of tractors with average power of 84 kW and 117 kW.

The agricultural land area (ALA) was located surrounding each farm, with fields close to it (farm-field distance equal to 10 m) and fields located farther (farm-field distance equal to 1500 m). According to this, an average farm-field distance equal to 1000 m was considered. Field shape was supposed to be rectangular, with a width-length ratio equal to 1:4 (identified shape per hectare: 50 m wide and 200 m long; slope was absent).

The coupling between tractor and implement was the most sensible considering the tractors available on farm and the engine power requests. The coupling choices as well as inventory data (i.e. implement working depth and width, worked days, working time, effective field capacity, fuel consumption and exhaust gases emissions and materials and lubricants consumed) are reported in the Supplementary material (Tables S1–S3).

With LCA, the extraction, production, transport, use, recycling and disposal of the inputs is also considered within the system boundary. These background data of raw material extraction (e.g., fossil fuels and minerals), manufacture (e.g., tractors and implements), use, maintenance and final disposal of machinery as well as of buildings for machinery shelter were obtained from Ecoinvent Database v3 (Althaus et al., 2007; Frischknecht et al., 2007; Jungbluth et al., 2007; Nemecek et al., 2015; Spielmann et al., 2007).

2.4.1. ENVIAM description

ENVIAM (ENVironmental Inventory of Agricultural Machinery operations) is a tool developed to support the filling of inventories for agricultural machinery operations characterised by the possibility of introducing local variables. In particular, ENVIAM consists of:

- a database of tractors and one of implements that are filled in with information from the market and from which the user selects the machinery to study;
- a section devoted to the introduction of input data (e.g., worked area and working time split in effective work, turns at headlands, transport, maintenance, etc.) and to the selection of local variables;
- a section for calculations necessary for coupling tractor and implement. Here, it is possible to pursue results adopting both the selected/available implement and tractor as well as the tractor that best responds to the power coupling calculations;
- a results section in which fuel consumption (kg/ha), lubricant consumption (g/ha), engine exhaust gases emissions (CO_2, CO, NO_X, HC, PM) (g/ha)[1] and materials depletion[2] (kg/ha) are quantified and available as inventory data.

Additional details about this tool can be found in Lovarelli et al. (2016, 2017).

2.5. Life Cycle Impact Assessment

For the impact assessment phase, the ReCiPe[3] midpoint method (Goedkoop et al., 2009) was adopted. It is a method used to transform the inventory data (inputs and outputs) in environmental impacts through characterisation factors. In other words, within the characterisation method, characterisation factors are numerical values used to attribute a share of environmental impact to one or more substances at a midpoint level (identified by means of impact categories). In this study, the following 13 impact categories were evaluated: climate change (CC; kg CO_2 eq), ozone depletion (OD; kg CFC-11 eq), terrestrial acidification (TA; kg SO_2 eq),

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[1] Quantified in accordance with the EU Directive 97/68/EC (and following amending ones: Directive 2010/26/EU, Directive 2010/22/EU) on the Emissive Stage.

[2] Mass of tractor and implement depleted during the operation.

[3] The ReCiPe method for LCIA has been given this name, because it provides a recipe to calculate life cycle impact category indicators. The acronym also represents the institutes' initials that mainly took part in its design.

Table 3
Sequence of operations per code (ploughing depth and soil refinement) and per soil texture. The same letters characterise the same sequence of operations. The codes identify: L_{25} = ploughing depth 25 cm and low refinement intensity; H_{25} = ploughing depth 25 cm and high refinement intensity; H_{35} = ploughing depth 35 cm and high refinement intensity. "N. rep" stands for the number of repetitions carried out with the implement.

Soil texture	Code								
	L_{25}			H_{25}			H_{35}		
	Operation		N. rep	Operation		N. rep	Operation		N. rep
Coarse texture	A	Mouldboard plough	1	A	Mouldboard plough	1	A	Mouldboard plough	1
		Spring tine harrow	1		Spring tine harrow	1		Spring tine harrow	1
	B	Mouldboard plough	1	B	Mouldboard plough	1	B	Mouldboard plough	1
		Fixed teeth harrow	1		Fixed teeth harrow	1		Fixed teeth harrow	1
	C	Disc plough	1	C	Disc plough	1	C	Disc plough	1
		Spring tine harrow	1		Spring tine harrow	1		Spring tine harrow	1
	D	Subsoiler	1	D	Subsoiler	1	D	Subsoiler	1
		Spring tine harrow	1		Spring tine harrow	1		Spring tine harrow	1
Medium texture	A	Mouldboard plough	1	A	Mouldboard plough	1	A	Mouldboard plough	1
		Spring tine harrow	1		Spring tine harrow	1		Spring tine harrow	1
	C	Disc plough	1	C	Disc plough	1	C	Disc plough	1
		Spring tine harrow	1		Spring tine harrow	2		Spring tine harrow	2
	D	Subsoiler	1	D	Subsoiler	1	D	Subsoiler	1
		Spring tine harrow	2		Spring tine harrow	3		Spring tine harrow	3
	E	Mouldboard plough	1	E	Mouldboard plough	1	E	Mouldboard plough	1
		Rotary harrow	1		Rotary harrow	2		Rotary harrow	2
	F	Mouldboard plough	1	F	Mouldboard plough	1	F	Mouldboard plough	1
		Disc harrow	1		Disc harrow	2		Disc harrow	2
	–	–	–	G	Subsoiler	1	G	Subsoiler	1
					Rotary tiller	1		Rotary tiller	1
	H	Subsoiler	1	H	Subsoiler	1	H	Subsoiler	1
		Disc harrow	2		Disc harrow	3		Disc harrow	3
	–	–	–	I	Disc plough	1	I	Disc plough	1
					Rotary harrow	1		Rotary harrow	1
	–	–	–	J	Mouldboard plough	1	J	Mouldboard plough	1
					Spring tine harrow	1		Spring tine harrow	1
					Rotary harrow	1		Rotary harrow	1
	–	–	–	K	Slatted plough	1	K	Slatted plough	1
					Disc harrow	1		Disc harrow	1
					Spring tine harrow	1		Spring tine harrow	1
Fine texture	A	Mouldboard plough	1	A	Mouldboard plough	1	A	Mouldboard plough	1
		Spring tine harrow	2		Spring tine harrow	3		Spring tine harrow	3
	E	Mouldboard plough	1	E	Mouldboard plough	1	E	Mouldboard plough	1
		Rotary harrow	1		Rotary harrow	2		Rotary harrow	2
	G	Subsoiler	1	–	–	–	–	–	–
		Rotary tiller	1						
	H	Subsoiler	1	H	Subsoiler	1	H	Subsoiler	1
		Disc harrow	2		Disc harrow	3		Disc harrow	3
	J	Mouldboard plough	1	J	Mouldboard plough	1	J	Mouldboard plough	1
		Spring tine harrow	1		Spring tine harrow	2		Spring tine harrow	2
		Rotary harrow	1		Rotary harrow	1		Rotary harrow	1
	K	Slatted plough	1	K	Slatted plough	1	K	Slatted plough	1
		Disc harrow	1		Disc harrow	1		Disc harrow	1
		Spring tine harrow	1		Spring tine harrow	2		Spring tine harrow	2
	–	–	–	L	Subsoiler	1	L	Subsoiler	1
					Rotary tiller	1		Rotary tiller	1
					Spring tine harrow	1		Spring tine harrow	1
	–	–	–	M	Mouldboard plough	1	M	Mouldboard plough	1
					Disc harrow	2		Disc harrow	2
					Rotary harrow	1		Rotary harrow	1

Notes: For details about implements, tractors and effective field capacity see Tables S1, S2, S3 in Supplementary material.

Table 4
Inventory of the selected tractors for farms F_A, F_M and F_F.

Farm	Tractor	Power kW	Mass kg	Annual working time h/yr
F_A	1	45	2992	500
	2	77	3841	600
	3	83	5240	300
F_M	1	64	4327	550
	2	97	5670	650
	3	113	7635	350
F_F	1	64	4327	550
	2	147	8225	700
	3	170	9707	400

freshwater eutrophication (FE; kg P eq), marine eutrophication (ME; kg N eq), human toxicity (HT; kg 1,4-DB eq), photochemical oxidant formation (POF; kg NMVOC), particulate matter formation (PM; kg PM10 eq), terrestrial ecotoxicity (TEx; kg 1,4-DB eq), freshwater ecotoxicity (FEx; kg 1,4-DB eq), marine ecotoxicity (MEx; kg 1,4-DB eq), metal depletion (MD; kg Fe eq) and fossil depletion (FD; kg oil eq).

3. Results

The environmental outcomes are reported per soil texture category. The figures present the evaluated sequences of operations; when a sequence was missing, it was not reported graphically.

The results involve, for all three soil texture categories, a section in which the impacts on all impact categories are quantified and discussed, and a second part in which the hotspot processes[4] are identified.

3.1. Coarse textured soil

The comparison among the alternative sequences studied for the coarse textured soil is illustrated in Fig. 2. A good condition of soil refinement is easily obtained with a single secondary tillage operation; therefore, the results are shown for the cases with different ploughing depth (i.e. H_{35} and H_{25}-L_{25}). H_{25} and L_{25} are put together, because in this case – due to the soil texture category – it is not possible to distinguish the highly refined from the lowly refined case. Fig. 2-top illustrates H_{25}-L_{25} for all evaluated impact categories. Option C (disc plough and spring tine harrow) is the most impacting, mainly due to disc ploughing that has higher fuel consumption (14.3 kg/ha) and higher tractor mass (5240 kg) that is subject to material depletion if compared with the mouldboard plough. For the complete seedbed preparation, fuel consumption is 15.9 kg/ha (14.3 kg/ha for disc ploughing + 1.6 kg/ha for harrowing). The least impacting sequence for all impact categories is minimum tillage (option D: subsoiler and spring tine harrow) where the impact ranges between −37% and −50% compared with the highest impacting option.

For these seedbed preparation alternatives, ploughing is the key process on all impact categories. It ranges between 69% and 79% of the total on FE, HT, TEx, FEx, MEx and MD and between 80% and 91% on CC, OD, TA, ME, POF, PM and FD. For solution D (no ploughing), the subsoiler is the main contributor (61–62% of the total impact for seedbed preparation for TEx, FEx and MEx and 89–96% for all remaining categories).

As regard to the case of H_{35} in the coarse textured soil (Fig. 2-bottom), disc plough and spring tine harrow (option C) show the highest environmental impact for all evaluated categories, and subsoiler and spring tine harrow (solution D) the lowest. Compared to option C, the impact reduction for option D is overall consistent (range between 41% on MD and 55% on ME). These differences are due to primary tillage: subsoiling has −42% fuel consumption in comparison with mouldboard ploughing (9.0 kg/ha and 15.6 kg/ha, respectively) and −45% of implement mass subject to wear. Mainly because of machinery mass and fuel consumption, primary tillage is the environmental hotspot; for options A, B and C ploughing contributes between 70% and 81% of the total impact on FE, HT, TEx, FEx, MEx and MD and 84%-92% on CC, OD, TA, ME, POF, PM and FD. Instead, in the no-plough option D, subsoiling is the hotspot (range between 61% on TEx, FEx and MEx and 86% on POF).

From the results, it emerges that disc plough has a higher environmental impact than mouldboard: fuel consumption, exhaust gases emissions and tractor wear are higher in Option C than in the other ones. However, the absence of ploughing in Option D shows that the environmental impact can be reduced considerably, although subsoiling still demands for medium-high tractor power. Moreover, constantly along the analysed options, deeper ploughing is the main variable affecting the increase in environmental impacts between L_{25}-H_{25} and H_{35}.

3.2. Medium texture soil

On the medium textured soil, the eventual need for more harrowing repetitions and the possibility of carrying out secondary tillage with two different implements complicates the results. In L_{25} (Fig. 3-top), Option E (mouldboard plough and rotary harrow) has the worst outcomes on CC, OD, TEx and FD because this sequence shows the highest fuel consumption (21.0 and 8.8 kg/ha, respectively). On the remaining 9 of 13 categories, minimum tillage Option H (subsoiler and disc harrow) has the highest environmental impact. On the opposite, the most preferable option for all impact categories except for TEx (for which it is the second best) is Option A (mouldboard plough and spring tine harrow) with an impact reduction ranging around 19% for CC, OD and FD and 36–46% for all other categories compared to the worst options.

Concerning the hotspot processes, ploughing is the main contributor to the environmental impact of all sequences in a range of 59%-89% of the total impact on all categories; the lowest values occur when rotary and disc harrows are present, since their environmental impacts have a more important role if compared to spring tine harrows. With minimum tillage (Option D and Option H) the main contributor to the environmental load is the subsoiler (55–84% and 40–70% in Option D and Option H, respectively). On FE, TEx, MEx and MD, however, the hotspot is secondary tillage (disc harrow repeated twice in Option H). Therefore, contrarily to the coarse textured soil, minimum tillage on medium texture soils does not show the lowest environmental impact, because carrying out twice the harrowing causes a higher environmental load. In particular, inputs production and consumption (machinery materials and fuel and lubricant consumption) are the main causes for a higher environmental burden.

In the group of H_{25} performed on the medium textured soil (Fig. 3-middle), the environmental impact is the highest on: (i) CC, OD, TA, ME, POF, PM and FD for Option G; (ii) FE, HT, FEx, MEx and MD for Option H and (iii) TEx for Option E. As concerns the first one, rotary tiller is the implement that mostly affects the impact categories mainly related to fuel consumption and exhaust gases emissions. In Option E and H, instead, the environmental burden is due to the number of repetitions of secondary soil tillage, which affects the materials consumed. For all impact categories, Option A has the best environmental behaviour (range between −36% on CC and −58% on FD), due to the very low impact caused by secondary soil tillage.

As regard to H_{35} (Fig. 3-bottom), Option E is the most impacting for impact categories affected by fuel consumption and exhaust gases emissions (CC, OD, TA, PM, TEx and FD), while Options H and G for those affected by materials consumption (FE, HT, FEx, MEx, MD in Option H and ME and POF in Option G). Instead, Option K (slatted plough, disc harrow and spring tine harrow) has the lowest environmental load on OD, TA, ME, POF and PM. For all remaining categories, Option A (mouldboard plough and spring tine harrow) has, once more, the most preferable performance. Slatted plough shows a lower impact than mouldboard, and also spring tine harrow is one of the least impacting operations (3.0 kg/ha of fuel consumed, 275 kg the implement mass, 1.85 ha/h the effective field capacity). In fact, PTO-implements involve higher fuel consumption and machinery wear than no-PTO-implements, which deeply influences the environmental burdens.

Similarly to the previous results, either ploughing or subsoiling (in minimum tillage options) are hotspot.

3.3. Fine textured soil

Fig. 4-top shows the results for L_{25} of the fine textured soil. Option G (mouldboard plough, rotary and spring tine harrows) has the highest environmental impact on FE, HT, TEx, FEx, MEx and MD due to the depletion of materials of three implements. On CC, OD, TA, ME, POF, PM and FD impact categories, Option H (subsoiler and rotary tiller) has the most detrimental environmental burden given the high fuel consumption and exhaust gases emissions. On the opposite, the best environmental performances are obtained with Option H (on MD), J (on CC, OD, TEx and FD) and K (TA, FE, ME, HT, POF, PM, FEx and MEx). Compared with the most impacting options, the best ones achieve reductions between −23% and −46% of the total impact for the seedbed preparation on all impact categories.

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[4] A process is considered hotspot when it is the prominent process responsible for the environmental impact. In every impact category, a hotspot process can be identified. The analysis of the hotspot processes is evaluated putting the environmental impact category = 100% and attributing a share of this to each included process.

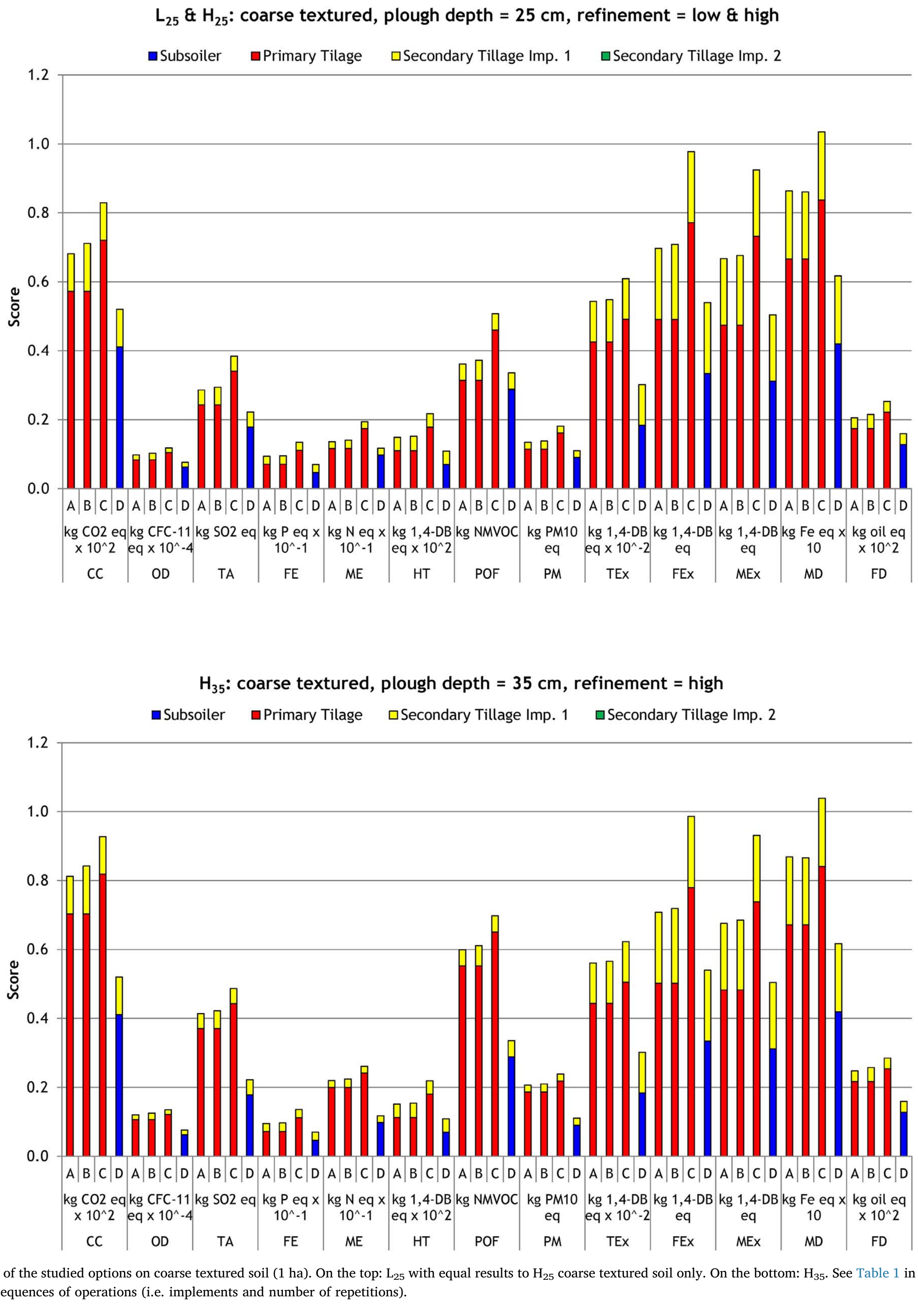


Fig. 2. Results of the studied options on coarse textured soil (1 ha). On the top: L_{25} with equal results to H_{25} coarse textured soil only. On the bottom: H_{35}. See Table 1 in Section 2.1 for details about sequences of operations (i.e. implements and number of repetitions).

In H_{25} (Fig. 4-middle), Option L (subsoiler, rotary tiller and spring tine harrow) has the highest environmental burden on CC, OD, TA, ME, POF, PM and FD impact categories, which is mainly caused by the rotary tiller (38.1 kg/ha fuel consumed and 0.52 ha/h effective field capacity). Instead, the lowest environmental impacts are, respectively: for Option E (mouldboard plough and rotary harrow) on TA, for Option J (subsoiler and disc harrow) on OD and TEx, for Option K (slatted plough, disc and spring tine harrow) on CC, FE, ME, HT, POF, PM, FEx, MEx and FD and for Option L on MD.

In H_{35} (Fig. 4-bottom), Option E (mouldboard plough and rotary harrow) has the highest environmental impact on CC, OD, TA, ME, POF, PM and FD impact categories, whereas Option H (mouldboard plough, rotary harrow and spring tine harrow) has the most detrimental effect on HT, TEx, MEx ad FEx and Option M on FE and MD. Option J (subsoiler and disc harrow) and K (slatted plough, disc harrow and spring tine harrow) are the best performing solutions on all evaluated impact categories.

Primary tillage, once more, represents the key process on all categories and for all sequences: in minimum tillage options the subsoiler is responsible for 33%-70% of the total impact, while in the conventional tillage options plough/rotary tiller for 44%-84% of the total impact.

4. Discussion

Other studies previously carried out on the environmental impact of crop production reported a high environmental responsibility attributable to soil tillage in a range between 26%-46% of the total impact of the different produced crops (Bacenetti et al., 2015b; Fedele et al., 2014; Fusi et al., 2014; Niero et al., 2015; Noya et al., 2015). Nevertheless, none of these studies aimed to compare alternative sequences of operations for seedbed preparation; moreover, since they used different functional units, system boundaries and Life Cycle Impact Assessment (LCIA) methods, the environmental results cannot be compared.

Mileusnić et al. (2010) built a model for fuel consumption for arable crop production. Similarly to this study, they assumed mechanical farm operative information (number of tractors, working promptness) and compared soil tillage alternatives (conventional, conservation and zero tillage). Decreasing fuel consumption (correct coupling, reduced tillage, etc.) is fundamental for reducing production costs and greenhouse gases emissions (Šarauskis et al., 2014).

As expected, in this study, for all the assessed alternatives the environmental impact was the lowest on coarse textured soils and highest on fine textured soils; moreover, it was the lowest for shallow ploughing and low refinement (L_{25}) and the highest for deep ploughing and high refinement (H_{35}). Working time and speed and the worked days per year affected the results because of both the duration of the operation and the farm-field distance and the related transport phases. Besides, fuel consumption and exhaust gases emissions are subjected to serious specific increases when the engine load (i.e. share of power used to the maximum power of the tractor) does not comply with optimal mechanical conditions (Bacenetti et al., 2015b; Lovarelli et al., 2017). Transport from farm to field and vice versa, similarly to turns at headlands, are characterised by a specifically high environmental load if compared to effective work, due to low engine load and, mainly, to transient conditions in engine (i.e. sudden change in engine speed and torque) (Lindgren and Hansson, 2004; Lovarelli et al., 2017; Pitla et al., 2016). Furthermore, field shape ratio (i.e. width and length) as well as implement working width influence the number of turns at headlands, which have a role on the total working time and environmental impact of the operation. For example, 3-m wide instead of 2-m wide harrows determine that only 60%-65% of turns at headlands must be completed, which reduces the working time and the related fuel consumption and exhaust gases emissions.

From the results has emerged that lower environmental impacts can be obtained when:

(i) low-energy-consumptive implements are used (e.g., no PTO harrows),
(ii) a lower number of implements are used (e.g., one for primary and one for secondary tillage and minimum tillage),
(iii) implements are characterised by low masses (e.g., slatted plough instead of mouldboard plough).

As regard to the number of implements used and their mass, lower impacts occur mainly for those categories affected by material production, consumption and depletion (e.g., TA, TEx, FEx, MEx and MD) as lower amounts of materials and energy are used, and fewer emissions are released to air, soil and water. Indeed, implements are often underused along their life span and better exploiting few of them would be less environmentally detrimental than using a wider fleet of implements for a shorter working time. This is even more valid when secondary tillage is carried out by no-PTO harrows, characterised by light masses (e.g., 200–300 kg), high field capacity (e.g., 1.5–1.8 ha/h) and low fuel consumption (e.g., 2.0-5.0 kg/ha). With these implements, impact

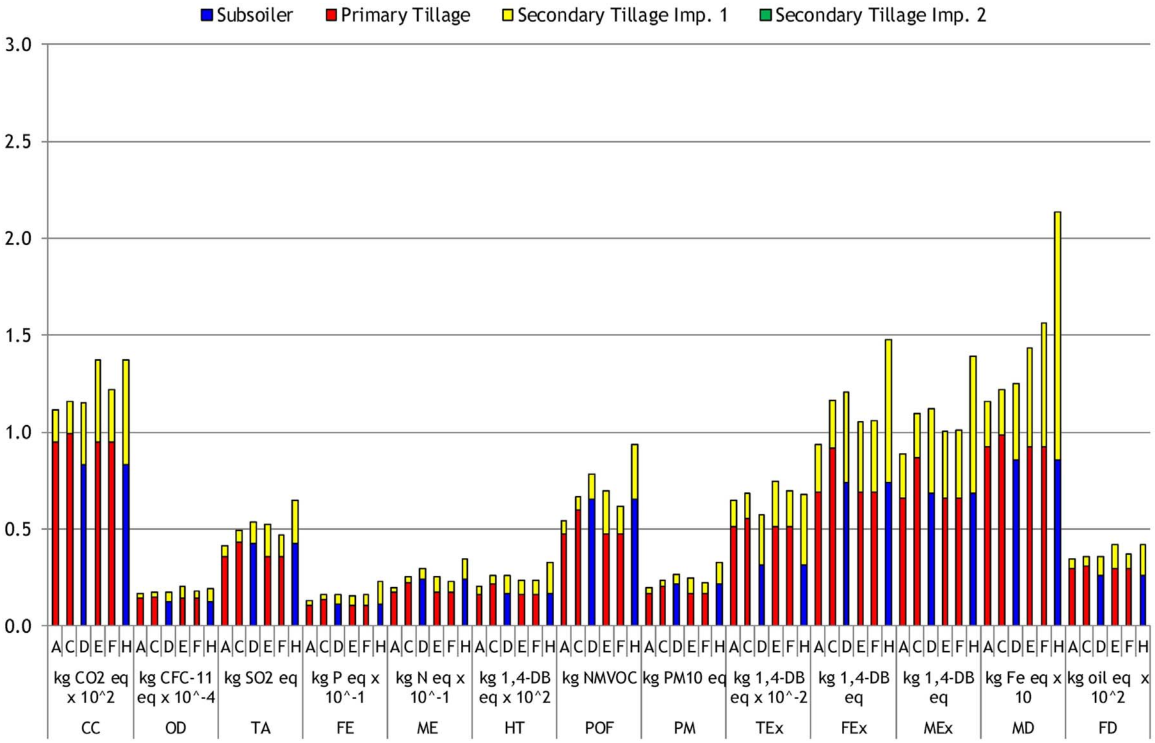


Fig. 3. Results of the studied options on medium texture soil (1 ha). On the top: L_{25}. In the middle: H_{25}. On the bottom: H_{35}. See Table 1 in Section 2.1 for details about sequences of operations (i.e. implements and number of repetitions).

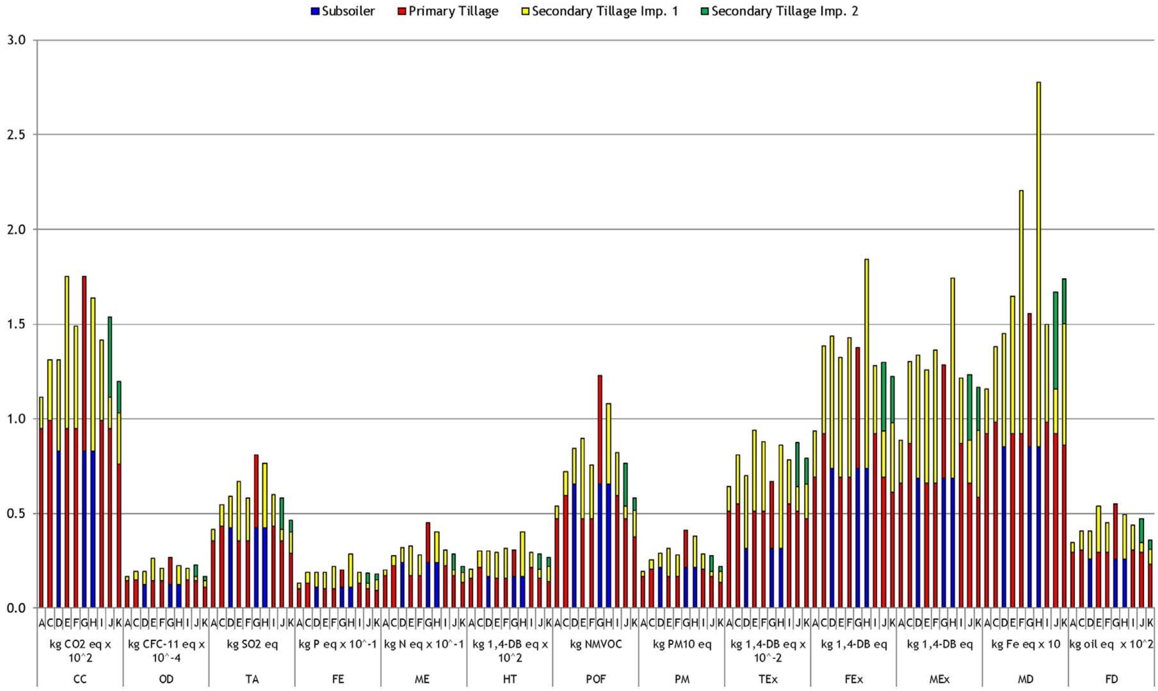
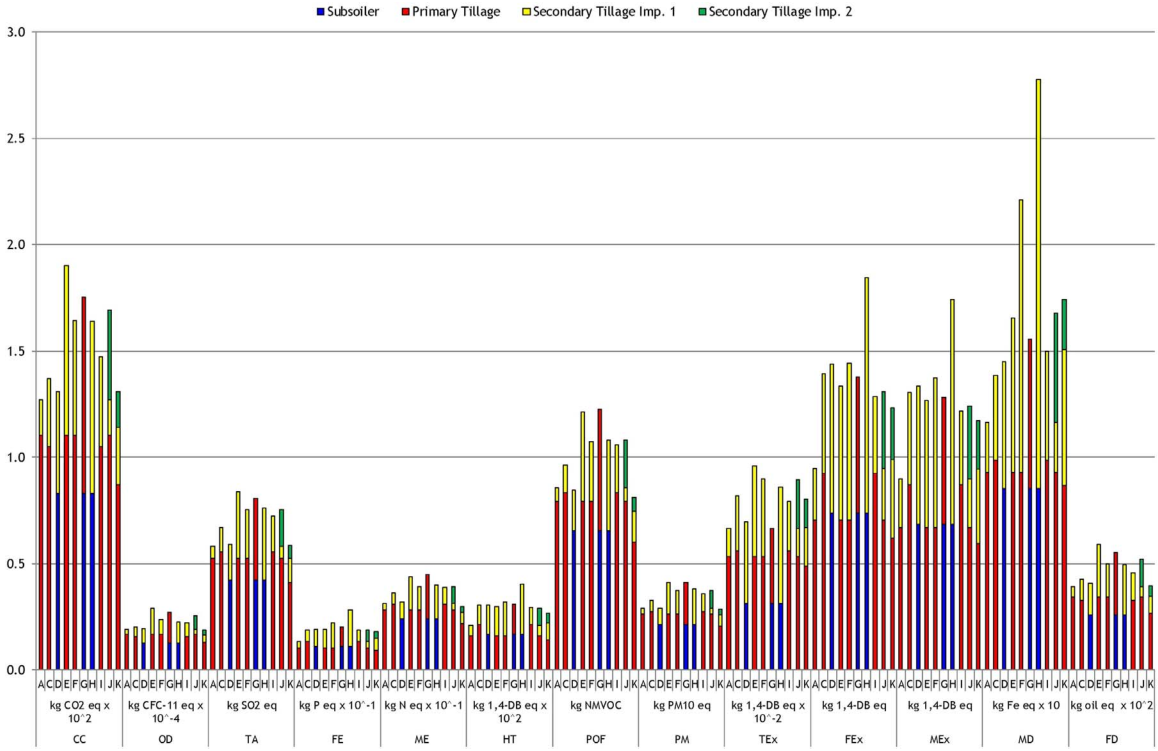


Fig. 3. (*continued*)



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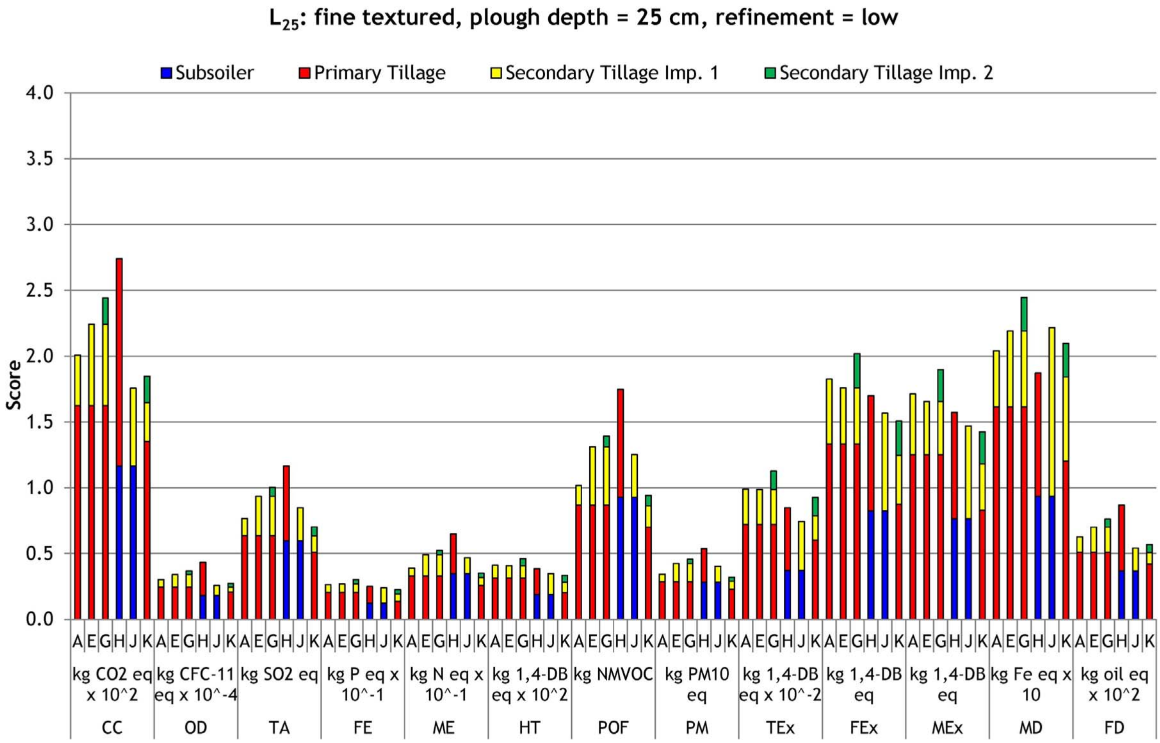
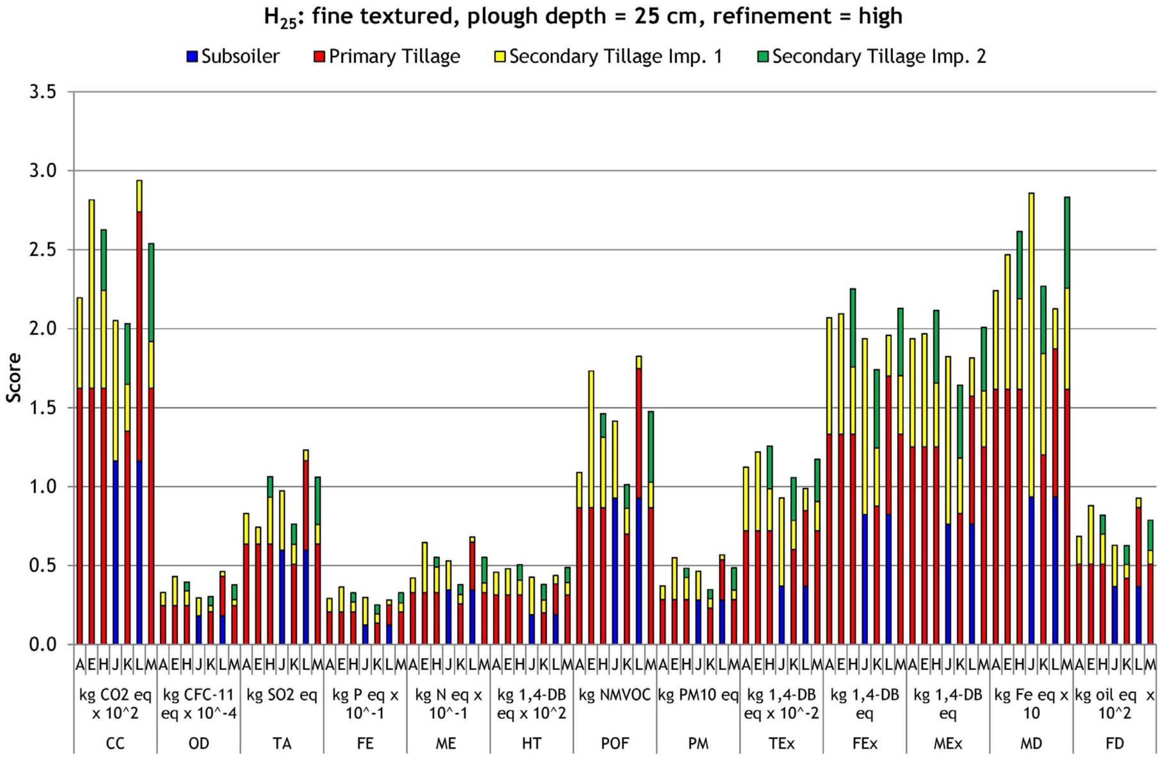


Fig. 4. Results of the studied solutions on fine textured soil (1 ha). On the top: L_{25}. In the middle: H_{25}. On the bottom: H_{35}. See Table 1 in Section 2.1 for details about sequences of operations (i.e. implements and number of repetitions).



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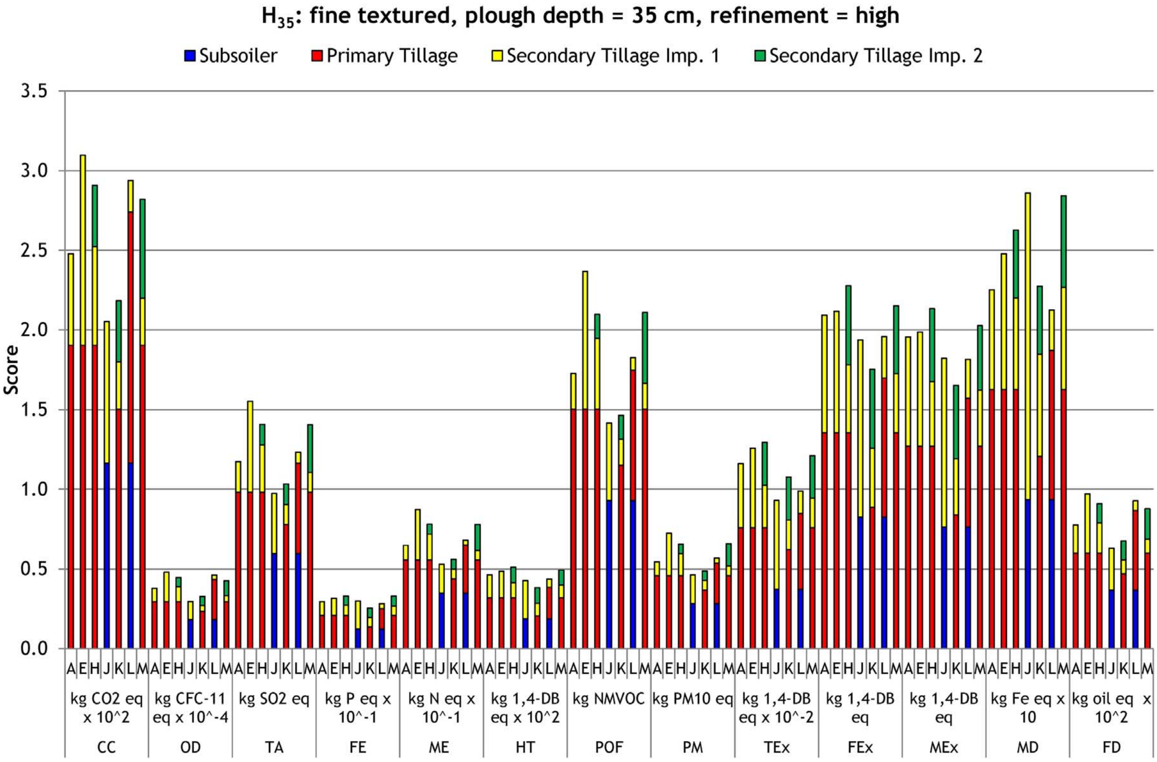

Fig. 4. (*continued*)

categories affected by fuel consumption and exhaust gases emissions (e.g., CC, OD and FD) also show positive outcomes.

Nevertheless, specific ploughing depth and soil refinement intensity required by the crop must be accurately evaluated, since the best seed germination is the main goal for farmers. Therefore, comparisons should be carried out only within the same group of seedbed preparation characteristics. In fact, the impact of H_{35} is undeniably higher than L_{25} and H_{25} with equal soil texture, but choosing H_{35} instead of H_{25} or L_{25} is an upstream decision due to the cultivated crop. Therefore, only the alternative sequences studied in the same grouping can be interestingly compared to identify the sequence with a lower environmental impact.

For example, on CC, the most impacting solution on medium texture soil has an absolute impact equal to 190.3 kg CO_2 eq/ha, while the related alternative in L_{25} is 136.9 kg CO_2 eq/ha. The same trend occurs on coarse textured (92.7 and 82.9 kg CO_2 eq/ha for H_{35} and L_{25}, respectively) and on fine textured (369.3 and 281.5 kg CO_2 eq/ha for H_{35} and L_{25}, respectively) soils. However, considering the most environmentally beneficial solution in each grouping, the results are more interesting and show an absolute impact on CC equal to: (i) on medium texture, 111.6 kg CO_2 eq/ha, 111.7 kg CO_2 eq/ha and 127.0 kg CO_2 eq/ha, respectively on L_{25}, H_{25} and H_{35}; (ii) on coarse textured soils, 52.0 kg CO_2 eq/ha, 52.1 kg CO_2 eq/ha, respectively on L_{25}-H_{25} and H_{35}; (iii) on fine textured soils, 200.7 kg CO_2 eq/ha, 203.2 kg CO_2 eq/ha and 205.2 kg CO_2 eq/ha, respectively on L_{25}, H_{25} and H_{35}. This means that, compared to the most impacting sequences, the selection of the optimal sequence of operations permits to have reductions in the environmental impact of seedbed preparation.

Another consideration is that, if soil tillage is not carried out with optimal conditions for tilth soil, the mechanisation identified with the proposed sequences could remarkably underestimate the effective environmental impact of seedbed preparation. In fact, soil moisture affects, on one hand, the tractor traction power and, on the other hand, the number of repetitions of secondary tillage for soil refinement (Ahmadi and Mollazade, 2009; Barthelemy et al., 1992); thus, it has effects on the agronomic, environmental, economic and operative points of view. Considering primary tillage, slatted plough determines a lower environmental impact if compared to the mouldboard (Lovarelli et al., 2017), while disc plough shows, usually, a behaviour similar or slightly more beneficial than the traditional mouldboard. Mass of implements (810 kg and 1795 kg, respectively for disc and mouldboard) and fuel consumption (17.1–22.6 kg/ha for disc and 15.6–43.0 kg/ha for mouldboard, depending on soil texture categories) imply the major dissimilarities. Subsoiling represents a valid substitute to ploughing for minimum tillage systems (Qingjie et al., 2014); instead, when used to complete an extraordinary operation, its impact must be summed to the ordinary operations (i.e. primary and secondary soil tillage). The use of a subsoiler for minimum tillage must match with adequate additional implements to produce pre-sowing operations with an environmental benefit. In fact, two options are usually available to farmers: either the farmer works the soil with his available implements or he charges contractors with the seedbed preparation. In this last case, contractors commonly have high-power tractors to be coupled with the high-power-demanding implements specifically developed for minimum tillage operations. This implies that the inventory data and, consequently, the environmental impact related to seedbed preparation can be much different if completed by a farmer or by a contractor.

As concerns secondary soil tillage, the environmental burdens are much variable. The evaluated non-PTO harrows (e.g., spring tine, fixed teeth, disc harrows) have very low contributions because the tractor power they require is very low (on average 5–15 kW) and fuel consumption is restrained (on average 2.0–5.0 kg/ha). On the opposite, rotary harrows and rotary tillers are more effective for soil refinement, but greatly affect the environmental results (e.g., higher tractor power absorbed and fuel consumed, lower field work capacity).

For farmers, working promptness and operative suitability are very important (Lazzari and Mazzetto, 2005) since the period within which to act can be very short (e.g., few days). For this reason, as well as for reducing labour and production inputs costs, farmers pursue solutions that permit seedbed preparation with implements working at fast speed, with high field capacity and not demanding for many repetitions. Accordingly, for example, rotary tiller is less common on Italian farms devoted to crops cultivation in the lowlands (i.e. low field capacity). However, it can still be interesting in other countries such as

Greece, Turkey and Morocco (Dimanche and Hoogmoed, 2002; Vasileiadis et al., 2007). Nevertheless, although giving interesting results in terms of soil refinement, its use should be discouraged because of its serious environmental impact.

5. Conclusions

In this study, using the LCA approach, the environmental impact of 13 alternative mechanical sequences for seedbed preparation of arable crop with two different ploughing depths and two refinement intensities were evaluated and compared within the same working assumptions. The identified alternatives were referred to typical working conditions of farmers, and not taking into account contractors.

The results highlight the environmental impact of the different mechanical solutions. By choosing the proper mechanisation for seedbed preparation, the environmental load can be steeply reduced for those impact categories affected by fuel consumption and exhaust gases emissions (e.g., climate change, ozone depletion, fossil depletion). Moreover, if low energy-consumptive implements characterised by high field capacity are used, also a higher benefit arises for the categories affected by materials production, consumption and disposal (e.g., terrestrial acidification, human toxicity, terrestrial ecotoxicity, metal depletion) because their production requires minerals, materials and energy. Most of the solutions are affected by this condition, since implements are often underused during their life span, and materials must be disposed of.

The approach used was limited to evaluating 13 sequences of operations. The effect of the operations on soil health and soil quality was not considered, although soil quality indices represent an important environmental variable as well (Congreves et al., 2015; Mitchell et al., 2017). Regarding this aspect, further research is needed in order to include it in the LCA approach.

Moreover, none of the studied sequences can be selected without considering the specific growing crop and local soil and climatic conditions. In particular, the selected sequences aimed to equally prepare the soil for seed emergence, although in minimum tillage sequences operations can determine a lower seed emergence. Specific features such as crop, geographical position and soil texture are among the most influencing features that drive farmers to selecting a sequence of operations and, consequently, to an environmental impact. Assuming that Mediterranean countries are characterised by similar crop growing systems, due to their similar pedo-climatic characteristics, the achieved results can be useful, on a large scale and on a large variety of working conditions, for the development of policies and incentive systems able to promote more environmental sustainable decisions.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.still.2017.06.006.

References

Çarman, K., 1997. Effect of different tillage systems on soil properties and wheat yield in Middle Anatolia. Soil Tillage Res. 40, 201–207.

Šarauskis, E., Buragiene, S., Masilionyte, L., Romaneckas, K., Avižienyte, D., Sakalauskas, A., 2014. Energy balance: costs and CO2 analysis of tillage technologies in maize cultivation. Energy 69, 227–235.

Ahmadi, H., Mollazade, K., 2009. Effect of plowing depth and soil moisture content on reduced secondary tillage. Agric. Eng. Int. XI, 1–9.

Althaus, H.J., Chudacoff, M., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. In: Life Cycle Inventories of Chemicals. Ecoinvent report No. 8, v2.0 EMPA. Swiss Centre for Life Cycle Inventories. Dübendorf, Switzerland.

Alvarez, R., Steinbach, H.S., 2009. A review of the effects of tillage systems on some soil physical properties, water content, nitrate availability and crops yield in the Argentine Pampas. Soil Tillage Res. 104 (1), 1–15.

Bacenetti, J., Fusi, A., Negri, M., Fiala, M., 2015a. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Bacenetti, J., Lovarelli, D., Ingrao, C., Tricase, C., Negri, M., Fiala, M., 2015b. Assessment of the influence of energy density and feedstock transport distance on the environmental performance of methane from maize silages. Bioresour. Technol. 466–467, 1066–1077.

Bacenetti, J., Duca, D., Fusi, A., Negri, M., Fiala, M., 2015c. Mitigation strategies in the agro-food sector: the anaerobic digestion of tomato puree by-products. An Italian case study. Sci. Total Environ. 526, 88–97.

Barthelemy, P., Boisgontier, D., Lajoux, P., 1992. Choisir les outils de travail du sol. Institut technique des cereales et des fourrages. Eymond Ed.

Boone, L., Van linden, V., De Meester, S., Vandecasteele, B., Muylle, H., Roldán-Ruiz, I., Nemecek, T., Dewulf, J., 2016. Environmental life cycle assessment of grain maize production: an analysis of factors causing variability. Sci. Total Environ. 553, 551–564.

Calcante, A., Mazzetto, F., Fontanini, L., 2011. Verifica dei parametri del modello per il calcolo dei costi di manutenzione e riparazione dei trattori 4RM in Pianura Padana: primi risultati. In: Midterm Conference of Italian Agricultural Engineering Association. Belgirate, 22–24 September 2011.

Castanheira, E.G., Dias, A.C., Arroja, L., Amaro, R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. Agric. Syst. 103 (7), 498–507.

Congreves, K.A., Hayes, A., Verhallen, E.A., Van Eerd, L.L., 2015. Long-term impact of tillage and crop rotation on soil health at four temperate agroecosystems. Soil Tillage Res. 152, 17–28.

Dace, E., Muizniece, I., Blumberga, A., Kaczala, F., 2015. Searching for solutions to mitigate greenhouse gas emissions by agricultural policy decisions—application of system dynamics modeling for the case of Latvia. Sci. Total Environ. 527–528, 80–90.

De Vita, P., Di Paolo, E., Fecondo, G., Di Fonzo, N., Pisante, M., 2007. No-tillage and conventional tillage effects on durum wheat yield, grain quality and soil moisture content in southern Italy. Soil Tillage Res. 92 (1–2), 69–78.

Dimanche, P.H., Hoogmoed, W.B., 2002. Soil tillage and water infiltration in semi-arid Morocco: the role of surface and sub-surface soil conditions. Soil Tillage Res. 66 (1), 13–21.

European Directive 97/68/EC, 1998. On emissions from non-road mobile machinery. Off. J. Eur. Commun. L59, 1.

Fedele, A., Mazzi, A., Niero, M., Zuliani, F., Scipioni, A., 2014. Can the Life Cycle Assessment methodology be adopted to support a single farm on its environmental impacts forecast evaluation between conventional and organic production? An Italian case study. J. Clear Prod. 69, 49–59.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Humbert, S., Köllner, T., Loerincik, Y., Margni, M., Nemecek, T., 2007. In: Implementation of Life Cycle Impact Assessment Methods. Ecoinvent Report No. 3, v2.0. Swiss Centre for Life Cycle Inventorie. Dübendorf, Switzerland.

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.

Goedkoop, M., Heijungs, R., Huijbregts, M., 2009. ReCiPe 2008. A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level. PRé Consultants, University of Leiden, Radboud University Nijmegen, RIVM, Bilthoven, Amersfoort, Netherlands.

Gronle, A., Lux, G., Böhm, H., Schmidtke, K., Wild, M., Demmel, M., Brandhuber, R., Wilbois, K.-P., Hess, J., 2015. Effect of ploughing depth and mechanical soil loading on soil physical properties weed infestation, yield performance and grain quality in sole and intercrops of pea and oat in organic farming. Soil Tillage Res. 148, 59–73.

Hokazono, S., Hayashi, K., 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. J. Clean. Prod. 28, 101–112.

IPCC, Agriculture, Forestry and Other Land Use, 2006. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. In: Eggleston (Ed.), .

Kouwenhoven, J.K., Perdok, U.D., Boer, J., Oomen, G.J.M., 2002. Soil management by shallow mouldboard ploughing in The Netherlands. Soil Tillage Res. 65 (2), 125–139.

Lazzari, M., Mazzetto, F., 2005. Prontuario di meccanica e meccanizzazione. REDA Ed.

Lee, J.W., Kim, J.S., Kim, K.U., 2016. Computer simulations to maximise fuel efficiency and work performance of agricultural tractors in rotovating and ploughing operations. Biosyst. Eng. 142, 1–11.

Lindgren, M., Hansson, P.-A., 2004. Effect of transient conditions on exhaust emissions from two non-road diesel engines. Biosyst. Eng. 81 (1), 57–66.

Lovarelli, D., Bacenetti, J., 2017. Bridging the gap between reliable data collection and the environmental impact for mechanised field operations. Biosyst. Eng. 160, 109–123. http://dx.doi.org/10.1016/j.biosystemseng.2017.06.002.

Lovarelli, D., Bacenetti, J., Fiala, M., 2016. Life cycle inventories of agricultural machinery operations: a new tool. J. Agric. Eng. XLVII, 40–53.

Lovarelli, D., Bacenetti, J., Fiala, M., 2017. Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. J. Clean. Prod. 140, 479–491.

Marques Da Silva, J.R., Soares, J.M.C.N., Karlen, D.L., 2004. Implement and soil condition effects on tillage-induced erosion. Soil Tillage Res. 78 (2), 207–216.

Mileusnić, Z.I., Petrović, D.V., Dević, M.S., 2010. Comparison of tillage systems according to fuel consumption. Energy 35 (1), 221–228.

Mitchell, J.P., Shrestha, A., Mathesius, K., Scow, K.M., Southard, R.J., Haney, R.L., Schmidt, R., Munk, D.S., Horwath, W.R., 2017. Cover cropping and no-tillage improve soil health in an arid irrigated cropping system in California's San Joaquin Valley, USA. Soil Tillage Res. 165, 325–335.

Molari, G., Bellentani, L., Guarnieri, A., Walker, M., Sedoni, E., 2012. Performance of an agricultural tractor fitted with rubber tracks. Biosyst. Eng. 111 (1), 57–63.

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 Bioenergy 63, 321–329.

Negri, M., Bacenetti, J., Manfredini, A., Lovarelli, D., Fiala, M., Bocchi, S., 2014b. Evaluation of methane production from maize silage by harvest of different plant portions. Biomass Bioenergy 67, 339–346.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C., 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Eur. J. Agron. 65, 40–51.

Niero, M., Ingvordsen, C.H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M.F., Hauschild, M.Z., Jørgensen, R.B., 2015. Eco-efficient production of spring barley in a changed climate: a Life Cycle Assessment including primary data from future climate scenarios. Agric. Syst. 136, 46–60.

Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K., 2015. Life Cycle Assessment in the Agri-food Sector. Springerpp. 1–390.

Noya, I., González-García, S., Bacenetti, J., Arroja, L., Moreira, M.T., 2015. Comparative life cycle assessment of three representative feed cereals production in the Po Valley (Italy). J. Clean. Prod. 1–16.

Panettieri, M., Knicker, H., Berns, A.E., Murillo, J.M., Madejón, E., 2013. Moldboard plowing effects on soil aggregation and soil organic matter quality assessed by 13C CPMAS NMR and biochemical analyses. Agric. Ecosyst. Environ. 177, 48–57.

Perozzi, D., Mattetti, M., Molari, G., Sereni, E., 2016. Methodology to analyse farm tractor idling time. Biosyst. Eng. 148, 81–89.

Pitla, S.K., Luck, J.D., Werner, J., Lin, N., Shearer, S.A., 2016. In-field fuel use and load states of agricultural field machinery. Comput. Electron. Agric. 121, 290–300.

Qingjie, W., Caiyun, L., Hongwen, L., Jin, H., Sarker, K.K., Rasaily, R.G., Zhonghui, L., Xiaodong, Q., Hui, L., Mchugh, A.D.J., 2014. The effects of no-tillage with subsoiling on soil properties and maize yield: 12-Year experiment on alkaline soils of Northeast China. Soil Tillage Res. 137, 43–49.

Schmidt Rivera, X., Bacenetti, J., Fusi, A., Niero, M., 2017. The influence of fertiliser and pesticide emissions model on life cycle assessment of agricultural products: the case of Danish and Italian barley. Sci. Total Environ. 592, 745–757.

Solinas, S., Fazio, S., Seddaiu, G., Roggero, P.P., Deligios, P.A., Doro, L., Ledda, L., 2015. Environmental consequences of the conversion from traditional to energy cropping systems in a Mediterranean area. Eur. J. Agron. 70, 124–135.

Vakali, C., Zaller, J.G., Köpke, U., 2011. Reduced tillage effects on soil properties and growth of cereals and associated weeds under organic farming. Soil Tillage Res. 111 (2), 133–141.

Van Linden, V., Herman, L., 2014. A fuel consumption model for off-road use of mobile machinery in agriculture. Energy 77, 880–889.

Vasileiadis, V.P., Froud-Williams, R.J., Eleftherohorinos, I.G., 2007. Vertical distribution, size and composition of the weed seedbank under various tillage and herbicide treatments in a sequence of industrial crops. Weed Res. 47 (3), 222–230.

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. ILCD Handbook-Towards more sustainable production and consumption for a resource efficient Europe. JRC Reference Report.

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Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation

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ABSTRACT

The environmental impact of crop production is mainly related to fossil fuels consumption and to fertilisers application. Emissions arising from the spreading of organic and mineral fertilisers are important contributors for impact categories such as eutrophication and acidification. The choice of the fertilisers and of the spreading techniques as well as the crop residues management can deeply affect the environmental impact related to crop cultivation.

In this study, seven scenarios describing fertilising schemes characterised by different organic and mineral fertilisers and by different mechanisation were compared. The aim is to evaluate, using the Life Cycle Assessment (LCA) method, how the environmental performances of grain maize production were affected by these different fertilisers schemes. The study was carried out considering a cradle to farm gate perspective and 1 t grain maize was selected as functional unit. Inventory data were collected on a farm located in Po Valley (Northern Italy) during year 2013 and were processed using the composite method recommended by the International Reference Life Cycle Data System (ILCD). The compared scenarios involved organic and mineral fertiliser distribution and were: pig slurry incorporation after >3 days after spreading (BS), fast pig slurry incorporation within 2 h from spreading (AS1), direct soil injection of pig slurry (AS2), pig slurry incorporation (after >3 days) with straw collection (AS3), digestate spreading instead of pig slurry (after >3 days) (AS4), only mineral fertilisers (i.e. urea and superphosphate) distribution (AS5) and only mineral fertilisers (i.e. calcium ammonium nitrate and superphosphate) distribution (AS6).

The results were not univocal, since climate and soil conditions as well as physical and chemical fertiliser characteristics differently affected the environmental load, especially for particulate matter formation, terrestrial acidification and terrestrial eutrophication impact categories. AS1 and AS2 showed the most beneficial results for these impact categories (between −67% and −73% respect to worst scenario). AS6, on the opposite, showed the highest environmental impact for those impact categories mainly affected by energy and fossil fuel consumption (climate change, ozone depletion, human toxicity with carcinogenic effect, particulate matter, freshwater eutrophication, freshwater ecotoxicity and mineral, fossil and renewable resources depletion), categories on which AS3 and AS4 were the best solutions. AS3 was the most impacting for terrestrial acidification and eutrophication

A sensitivity analysis was carried out varying grain maize yield (mostly affected: marine eutrophication) and ammonia volatilisation losses due to organic fertilisers (mainly affected: terrestrial acidification and eutrophication).

The achieved results can be useful for the development of "spreading rules" that drive the application of organic fertilisers in agricultural areas where there is an intense livestock activity.

1. Introduction

Evaluation of the environmental performance associated with agricultural processes is increasingly becoming important because agriculture is responsible for remarkable environmental impacts.

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According to FAO, the greenhouse gases (GHG) emissions related to agricultural systems have increased of about 96% from 1961 to 2011 (from 2.7 to about 5.3 billion tonnes of CO_2 equivalents). Among the different emissions sources, enteric fermentation plays the major role, but also GHG emissions related to synthetic fertilisers and to manure management cannot be neglected. In Italy, in 2012, GHG emissions from agriculture were about 32,100 t and the most important emission sources were enteric fermentation (37.5%), manure management (20.0%), synthetic fertilisers (13.8%), emissions from manure application to the soil (8.9%) and rice cultivation (8.1%) (FAOSTAT, 2016). Above all, this occurs in Northern Italy, where the livestock activities are concentrated (66% of Italian cow livestock and 87% of Italian pig livestock; ISTAT, 2016) and there is an abundant availability of animal waste (manure and/or slurry) with important environmental concerns. In particular, from the Italian Agricultural Census of 2010 emerges that in Po Valley are produced about 11.0 and 55.2 million t year^{-1} of pig and cow slurries, respectively. Therefore, the slurry mass to manage is very high. The management of organic fertilisers is regulated by the European Nitrates Directive (91/676/EEC) and also by regional regulations and guidelines (e.g., LR n. 37/1993, D.g.r. 2244/2006, D.g.r. 10890/2009). Mainly, slurry is spread before summer crops sowing (e.g., maize) and the agricultural area dedicated to maize cultivation is about 752,220 ha (ISTAT, 2016), which is quite a limited area for the N application and regulation enforcement.

As regard to crops cultivation, the environmental impact is usually related to fossil fuels use, fertilisers production (Hasler et al., 2015) and emissions generated by the application of organic and mineral fertilisers (González-García et al., 2012; Nemecek et al., 2015; Wang and Dalal, 2015). In more details, when organic fertilisers are applied, emissions of nitrogen and phosphorous compounds take place (De Vries et al., 2015). The first (e.g., ammonia, dinitrogen oxide, nitrate) occur because of volatilisation, denitrification and leaching, while the second mainly because of run-off, but also leaching (Brentrup et al., 2001; Ciampitti et al., 2011; Maguire and Sims, 2002). In particular, ammonia emissions deeply affect soil acidification, water eutrophication and particulate matter formation (Bacenetti et al., 2015a; De Vries et al., 2015). Mostly, these emissions take place immediately after the application of organic fertilisers and are strictly dependent upon climatic and soil characteristics (e.g., temperature, rainfall, wind, soil texture and structure and pH) (Brentrup et al., 2004a,b), but are also deeply affected by spreading techniques. Fast soil incorporation as well as direct soil injection were recognised as spreading techniques capable to reduce considerably ammonia volatilisation phenomena (Carozzi et al., 2013; Ferrara et al., 2016; Minoli et al., 2015). Although there are evidences about their effectiveness for ammonia emission mitigations, there is no information about their impact on the environmental performances. Instead, there is a need of multicriteria assessment that addresses several environmental issues not focusing only on the reduction of ammonia emissions. In fact, the focus only on field emissions is not enough: the substitution of one fertiliser with another could reduce NH_3 emission, but in the meanwhile it could involve a higher energy consumption or higher emissions of P-compounds. For example, direct soil injection determines a substantial request of traction force, which causes increase of fuel consumption that could offset the environmental benefits arising from ammonia emissions reduction. Currently, there are no detailed studies on a global perspective that evaluate the environmental benefits that arise from adopting one spreading technique instead of another. The choice of the fertiliser as well as of the spreading technique must be evaluated with holistic assessment methods that permit to evaluate the whole crop cycle. Among the environmental assessment methods implemented over the years (Bockstaller et al., 2009; Gaudino et al., 2014; Lebacq et al., 2013) the Life Cycle Assessment (LCA) is the most adopted approach to fulfil this requirement. It is the most applied approach also thanks to the availability of specific ISO standards (ISO, 2006). By using LCA, it is possible to analyse the potential environmental impacts of products (processes or services) throughout their whole life cycle. The LCA has been used in several studies for assessing environmental impacts of fertilisers application (Brentrup et al., 2000; Brockmann et al., 2014; Hasler et al., 2015; IPCC, 2006). However, in this study the environmental impact of alternative mechanisation solutions for organic fertilising and different organic and mineral fertilisers were investigated, which brings new knowledge to the sector and permits to identify the environmental benefits and drawbacks of each spreading solution and selected fertiliser.

The aim of this study is to evaluate, using the LCA methodology with a cradle to farm gate perspective, how the environmental performances of grain maize production are affected by fertilising schemes characterised by different organic and mineral fertilisers as well as different organic fertilisers spreading techniques.

2. Materials and methods

Life cycle assessment (LCA) has been used to estimate the environmental impacts of maize grain production system, following the ISO 14040/44 methodology (ISO 14040, 2006) and the EPD guidelines developed for “Arable Crops” (Environdec, 2014).

In more details, the following sections are devoted to the four steps of LCA studies: (i) goal of the study, selection of the functional unit, description of the system and of the system boundary, (ii) Life Cycle Inventory data collection, (iii) Life Cycle Impact Assessment, (iv) interpretation of the results and identification of the process hotspots.

The LCA is performed with real data (see Sections 2.2–2.4) for a baseline scenario and for 6 alternative scenarios, as described in Section 2.3.1.

2.1. Goal of the study and selection of functional unit

The goal of this study is to evaluate the environmental benefits achievable from the adoption of different organic fertilisers spreading techniques. To this purpose, maize cultivation in Northern Italy and, in particular, in Lombardy region was considered.

The achieved results are useful for the development of “spreading rules” able to drive the application of organic fertilisers on agricultural areas where there is an intense livestock activity towards a more sustainable manure management.

The functional unit (FU) is defined as a quantified performance of a product system to be used as a reference unit in a LCA (ISO, 2006ISO 14040, 2006). Although different FU such as area can be used (Halberg et al., 2005; Nemecek et al., 2015; Solinas et al., 2015) the mass based FU is widely used for LCA of agricultural systems (Bacenetti et al., 2015b; Fedele et al., 2014; Notarnicola et al., 2015). In this study, 1 t of maize grain (at commercial moisture of 14%) was selected as FU because in the evaluated farm cereals are grown to produce grain to be sold.

2.2. Description of baseline scenario

The analysed crop was cultivated in the Po Valley area (Northern Italy) and more precisely on a farm in the district of Milan. The global cultivated area was 40 ha, whose 92% was managed under irrigation conditions. Grain yield was evaluated for 5 years, during the cultivation seasons 2011–2015. The local climate is characterised by an annual average temperature of 12.7 °C, ranging between an average of −1.0 °C in January and +23.2 °C in July. Rainfall is on average 795 mm per year, mainly distributed in spring and autumn. Soil was of medium texture on all fields, well-drained,

Table 1
Main information about maize grain cultivation practice.

Operation	Description
Organic fertilisation	Pig slurry spreading with slurry tanker in April
Organic fertiliser incorporation and primary soil tillage	Ploughing with 3 mouldboard ploughshares plough in April
Secondary soil tillage	Two harrowing interventions with rotary harrow in April
Sowing	Pneumatic precision seeder in April
Chemical weed control	With pre-emergence herbicide within 1–4 days after sowing and with post-emergence herbicide in May (until machinery can work on field without mechanical damages to the crop, around 4 leaves unfolded)
Mineral fertilisation	With urea in May (within the 4 leaves unfolded)
Mechanical weed control	Mechanical hoeing within few hours after mineral fertilisation
Irrigation	Pump coupled with tractor, 3 interventions in June–August
Harvesting	Combine harvester in September
Transport	Two farm trailers used to transport grain to the farm
Grain drying	Farm dryer using natural gas
Straw management	Straw chopped and incorporated into the soil the following spring

with slope <1% and pH ranging between 6.7 and 6.8 (Negri et al., 2014a,b).

Maize FAO Class 700 was sown in April and harvested in September, with a crop growing cycle equal to about 140 days. Requiring a long growing period that covers the soil from spring to late summer, maize FAO Class 700 does not allow for any other previous cultivation. This cropping system aimed to grain production is commonly a monoculture, which means that during winter season the soil is bare.

For what concerns the pre-sowing organic fertilisation, it was performed by means of pig slurry surface spreading. In more details, surface spreading was carried out with a slurry tanker (20.0 m^3) coupled with a 130 kW 4-wheel-drive (WD) tractor. For this operation, a precautionary traction force of 21 kN was required and tractor power was calculated considering a working speed equal to 5 km h^{-1} and a global tractor efficiency equal to 56% (Lazzari and Mazzetto, 2005).[1] During transfers on paved road, average speed was 25 km h^{-1}, which affected tractor power with a higher extent, although lower traction force (i.e. 7 kN). Considering also the loading time and the effective working width, the total working time for the operation was 2.6 h ha^{-1}. In total, 85 t ha^{-1} were distributed, on 8th April. On this day, the measured temperature was 17.0 °C (9 °C min; 21 °C max) and during the following 3 days no rainfall occurred and wind speed was lower than 5 km h^{-1}. During the month, the total precipitation was 65 mm. The organic fertiliser incorporation was carried out with ploughing after more than 72 h from spreading. Ploughing was carried out with a three ploughshares mouldboard plough coupled with a 90 kW 4WD tractor.

More details about the cultivation practice are reported in Table 1.

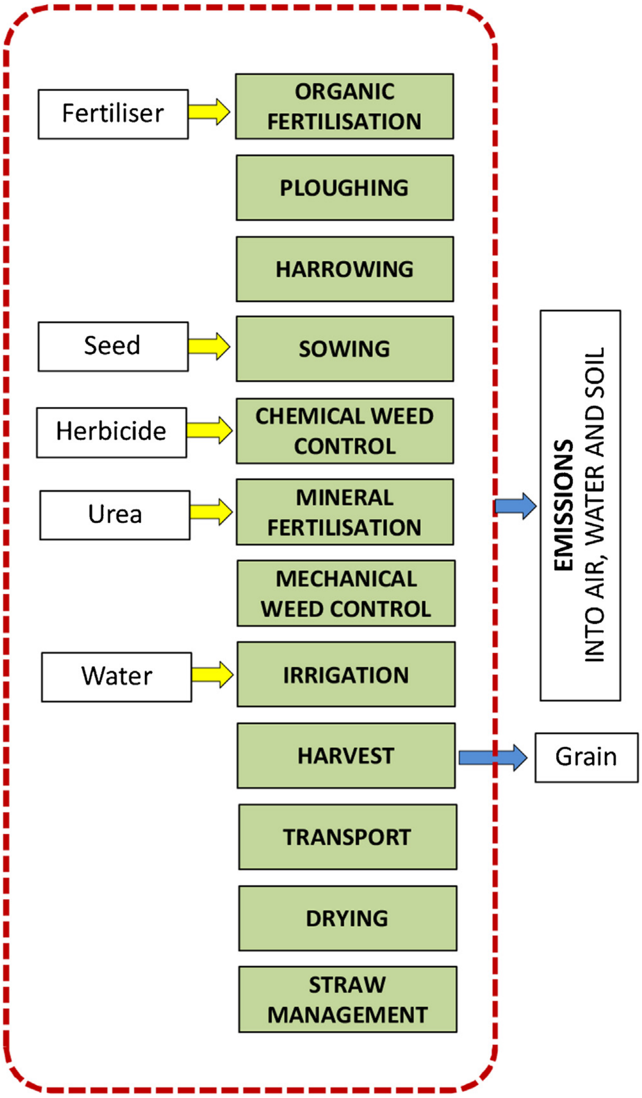

Fig. 1. System boundary for maize production: operations of the cultivation process (on green background) and inputs-outputs (on white background). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

2.3. System boundary

The system boundary for the cereal grain crop (baseline scenario—BS) is reported in Fig. 1. A cradle to farm gate perspective has been adopted; therefore, the system starts with the application of organic fertiliser and ends with grain drying and straw management. In more details, the following activities were included in the analysis: raw materials extraction (e.g., fossil fuels), manufacture of agricultural inputs (e.g., seed, fertilisers, pesticide and agricultural machines) and energy, use of the agricultural inputs (fertilisers emissions, pesticide emissions, diesel fuel emissions and tire abrasion emissions), maintenance and final disposal of machines and supply of inputs to the farm.

According to previous studies (González-García et al., 2012; Niero et al., 2015), no environmental load was considered for the organic fertilisers used, since they are waste of livestock activities.

Among the capital goods involved in the production process, tractors, equipment and the dryer were included, while other infrastructures (e.g., building) were excluded.

The agricultural area surrounding Milan, where the fields under study were located, was cultivated with cereals crops for many

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[1] Traction force (F_{tr}; N) is calculated as: $F_{tr} = m_{OM} \times 9.81$ N $kg^{-1} \times c_r$, where m_{OM} is the implement mass (kg) and c_r the rolling coefficient (no unit). Tractor power (Pm; kW) is calculated as: $Pm = (F_{tr} \times v_a)/\ {}_g$, where v_a is the working speed (km h^{-1}) and ${}_g$ the global tractor efficiency. A surplus of 20% of power is also considered to overcome possible harder working conditions.

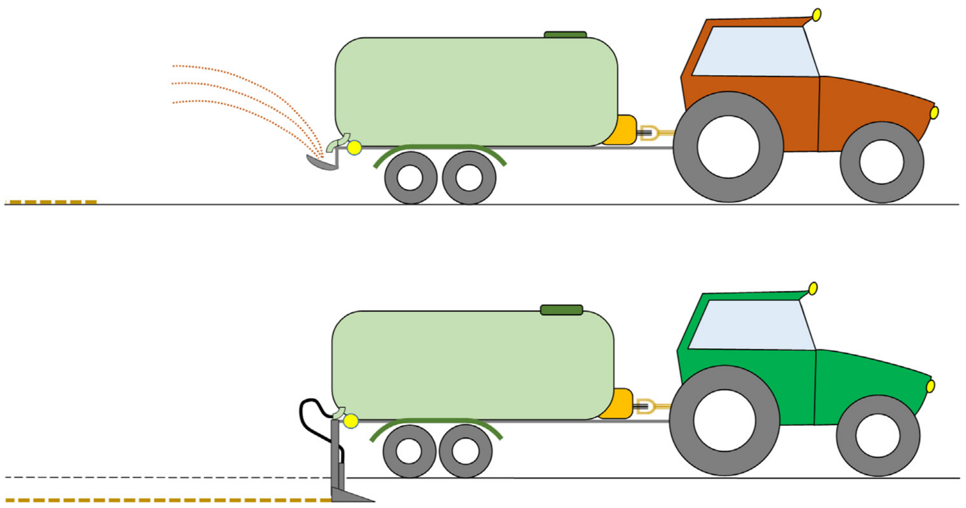

Fig. 2. Different application techniques for organic fertilisers. On the top: surface spreading with a slurry tanker, after which soil incorporation during ploughing needs to be carried out. On the bottom: direct soil injection of slurry carried out with a slurry tanker equipped with 5 anchors.

years (>30 years) and the soil carbon content was supposed to be in equilibrium. Therefore, carbon sequestration into the soil was not included, following the recommendations of PCR "arable crops" (Environdec, 2014).

2.3.1. *Alternative scenarios*

Beside BS (Fig. 2—top), six alternative scenarios were analysed:

- AS1—"fast incorporation", this scenario involved pig slurry surface spreading and incorporation into the soil within 2 h after the slurry application. The same mechanisation foreseen for BS was adopted;
- AS2—"soil injection", where a different technique was considered: direct injection of pig slurry into the soil. Direct injection was performed with a slurry tanker (20.0 m^3) equipped with 5 anchors working at 7–8 cm depth that directly injected pig slurry into the soil. The tanker was coupled with a 180 kW 4WD tractor (Fig. 2—bottom) because of the higher traction force requested for the operation. Namely, the traction force was 33 kN, of which 12 kN were due to the injecting system.[2] The distribution flow of the direct soil injection system permits to increase working speed, which is 7 km h^{-1}. Tractor power was calculated considering 56% as tractor global efficiency. Working time was different from BS, with a total equal to 3.2 h ha^{-1} due to the lower working width;
- AS3—"straw collection", in this scenario the straw was baled and collected to be sold. Organic fertilisers were spread with a mechanisation scheme similar to BS. However, considering that straw collection involved an additional removal of nutrient from the soil (Fusi et al., 2014), a supplementary amount of pig slurry was spread;
- AS4—"digestate", where digestate from a biogas plant fed with maize silage and pig slurry was distributed instead of pig slurry, with the same mechanisation scheme available for BS;
- AS5—"only mineral-urea for N", in this scenario the fertilisation was carried out using only mineral fertilisers (urea and triple superphosphate) with a tractor coupled with a mineral fertilisers spreader working at 10 km h^{-1} and for a total working time of 0.5 h ha^{-1};
- AS6—"only mineral-CAN for N", this scenario was characterised by only mineral fertilisers and by an equal mechanisation scheme to AS5. However, calcium ammonium nitrate was spread instead of urea. Triple superphosphate was also applied with the same amount as AS5.

Table 2
Evaluated scenarios.

Scenario	Acronym	Fertilisers	Timing of soil incorporation for organic fertiliser
Baseline	BS	Pig slurry and urea[a]	>3 days after spreading
Fast incorporation	AS1		<2 h after spreading
Soil injection	AS2		During the spreading
Straw collection	AS3		>3 days after spreading
Digestate	AS4	Digestate and urea	>3 days after spreading
Only mineral (urea for N)	AS5	Urea and triple superphosphate	n/a
Only Mineral-(CAN for N)	AS6	Calcium ammonium nitrate and triple superphosphate	

[a] In AS3 respect to BS/AS1/AS2 an additional amount of pig slurry is spread to compensate the higher nutrient removal.

In all the scenarios where organic fertiliser schemes were studied, the same amount of urea was applied during top mineral fertilisation (60 kg ha^{-1} of urea equal to 27.6 kg ha^{-1} of N). Table 2 summarises the different scenarios evaluated, no change in grain yield was considered.

2.4. *Inventory data collection*

The main inventory data were collected on a farm located in the district of Milan (Italy) as described in Section 2.2.

Data about maize cultivation and storage (e.g., field operations sequence, fuel consumption, seed, fertilisers, agro-chemicals and water use) were collected during 2013 by means of surveys at the farm and farmer interviews and were representative for grain maize cultivation in irrigated fields.

In more details, information concerning field operations and drying (e.g., working times, characteristics of tractors and implements such as mass, age, power, length, width and life span) and

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[2] In addition to the traction force (F_{tr}; N) necessary for towing the implement, the traction force of the injecting system must be considered. It is calculated taking into account a coefficient (; N m^{-1} cm^{-1}) of 260 N m^{-1} cm^{-1}, 5 anchors working at 7 cm depth (H; cm) and an implement working width (b; m) of 2.5 m. Traction force for the injecting system is calculated as: $F_{tr} = \quad \times b \times H$.

the amount of production factors applied (e.g., fertiliser, water, etc.) were collected through a survey form.

The amount of tractors and implements needed for each field operation was calculated considering the annual working time and the physical[3] and economical[4] life span (Fiala and Bacenetti, 2012).

Table 3 reports the main inventory data for Baseline Scenario (BS), the characteristics of the employed tractors and implements as well as the amount of production factors applied.

The grain yield was measured by means of the farm weighbridge; a yield of 14.1 t ha^{-1} of fresh matter (equal to 12.3 t ha^{-1} with a moisture content of 14%) was considered as a medium of the yield recorded over the five growing seasons.

Field emissions into air, water and soil of nitrogen compounds were assessed according to the model EFE-So, (Estimation of Fertilisers Emissions-Software, available at: http://www.sustainable-systems.org.uk/tools.php) (Fusi and Bacenetti, 2014) that, similarly to the one proposed by Brentrup et al. (2000), considers the parameters reported in Table 4. The model EFE-So assesses the emission of ammonia (NH_3) and dinitrogen oxide (N_2O) in air and of nitrate (NO_3) and phosphate (PO_3) in water considering soil type, climatic conditions and agricultural management operations. Ammonia volatilisation from organic fertilisers application was assessed considering (i) air temperature, (ii) time between the application and the rainfall or the incorporation in the soil; (iii) infiltration rate according to the fertiliser application circumstances (e.g., presence of crop residues on the soil). NH_3 emission from mineral fertiliser application was evaluated taking into account (i) type of fertiliser, (ii) climatic conditions and soil properties (e.g., pH, texture). N_2O emissions were computed considering the emission factor proposed by the IPCC (IPCC, 2006).

Finally, NO_3 emissions into water due to leaching were estimated considering (i) nitrogen balance, (ii) field capacity in the effective rooting zone, (iii) rainfall, (iv) drainage water zone.

The nitrogen balance considers (i) supply of N, coming both from the application of mineral and organic fertilisers, as well as from the N released from crop residues mineralisation, and (ii) N removal from the harvested biomass. If during the crop cycle the nitrogen removal is higher than the N supply, no NO_3 leaching occurs. On the contrary, if the N applied is higher than the one removed and during the winter season (when the soil is bare due to no crop presence) the rainfall exceeds the field capacity in the effective rooting zone, leaching takes place.

For AS1 and AS2, the experimental data recorded by Carozzi et al. (2013) in the same geographic area were considered. In more details, for ammonia the following emission reductions were considered:

- 84% for AS1 “Fast incorporation”,
- 95% for AS2 “Injection”.

For AS3, straw yield was equal to 14.5 t ha^{-1} (Baldoni and Giardini, 2000) and the nutrient removal related to straw collection was assessed considering a nitrogen content in the biomass equal to 0.75% of fresh matter. This nitrogen removed was supposed to be replaced by additional application of pig slurry and urea in the same proportion than in BS. Consequently, 123 t ha^{-1} of pig slurry and 86 kg ha^{-1} of urea were applied.

For AS4, the amount of digestate was computed considering a mineral fertiliser equivalence (MFE)[5] of the N applied with the organic fertiliser equal to 75% (instead of the 60% considered for pig slurry) (Amon et al., 2006; Hamelin et al., 2014; Lijó et al., 2014b; Regione Emilia-Romagna, 2011; Vu et al., 2015; Wulf et al., 2006). Considering the average digestate composition, in AS4 56.4 t ha^{-1} of digestate were applied.[6] As for pig slurry, no environmental impact was taken into account for digestate.

In AS6 868 kg of calcium ammonium nitrate (CAN) were considered.

Phosphate emissions were calculated following Prahsun (2006) and Nemecek and Kägi (2007); in more details, two different phosphorus emissions into water were considered:

- leaching to ground water: assessed using a factor of 0.07 kg P ha^{1} year^{-1}; and
- run-off to surface water: evaluated considering 0.175 kg P ha^{-1} year^{-1} as emission factor.

Due to a lack of data about fraction of the eroded soil, phosphate emissions through erosion to surface waters were not included.

With regard to AS5, only mineral fertilisers were distributed, in the amounts of 500 kg ha^{-1} urea and 150 kg ha^{-1} triple superphosphate. Also in AS6, only mineral fertilisers were applied: 898 kg ha^{-1} CAN and 150 kg ha^{-1} triple superphosphate.

Table 5 highlights the main differences among BS and AS for what concerns the emissions from organic and mineral fertilisation.

Pesticide emissions into the environment were assessed considering the model proposed by Margni et al. (2002).

Background data for the production of seed, diesel fuel, fertilisers, pesticides, tractors and agricultural machines (equipment and combine harvester) were obtained from the Ecoinvent database Database v.3 (Althaus et al., 2007; Frischknecht et al., 2007; Jungbluth et al., 2007; Nemecek and Kägi, 2007; Spielmann et al., 2007). Table S1 (see Supplementary Table S1 in the online version at DOI: 10.1016/j.eja.2016.05.015) reports the different Ecoinvent processes considered in the analysis.

2.5. Allocation

Allocation of environmental burdens is a recognised methodological problem in LCA and it is needed when a system fulfils more than one function. With allocation the environmental burdens is shared among each functional input or output of a multiple-function system. The systems under assessment in this study can be considered as multiple-function system since they generate two marketable products (grain and straw).

Grain is the main product of maize cultivation. However, also straw is produced during cultivation. When straw is sold, the environmental burdens of maize cultivation must be shared between grain and straw by means of allocation procedure. In the Scenarios (BS, AS1, AS2, AS4, AS5 and AS6) where straw was chopped and incorporated into the soil, allocation was not necessary.

In AS3, following the guidelines for EPD in “arable crops”, an allocation based on economic criteria was performed considering the selling price of grain and straw. This resulted in the allocation factors displayed in Table 6.

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[3] Physical life span (PLS, h) was considered equal to 12,000 h for tractors, 2000 h for plough, harrow, seeder and organic and mineral fertiliser spreader, 2500 h for the self-propelled harvester and 3000 h for farm trailers (Bodria et al., 2006).

[4] Economical life span (ELS, years) was 12 years for tractors and farm trailers, 10 years for self-propelled harvester and mineral and organic fertiliser spreader; 8 years for plough, harrow and seeder.

[5] The mineral fertiliser equivalency (MFE) for nitrogen is a measure of the fertiliser ability to supply nitrogen to crops compared with mineral fertiliser

[6] In BS, 85.0 t ha^{-1} of pig slurry with a nitrogen content of 2.43 kg t^{-1} was applied. With a MFE of N in the slurry equal to 60%, the corresponding efficient nitrogen is 123.93 kg ha^{-1}. In AS4, 56.4 t ha^{-1} of digestate with a nitrogen content of 2.93 kg t^{-1} was applied. With a MFE of N in the digestate equal to 75%, the corresponding efficient nitrogen is 123.93 kg ha^{-1}.

Table 3
Main inventory data concerning the agricultural machinery operations present in the baseline scenario (BS, surface spreading of pig slurry and incorporation after >3 days).

Operations	Rep.[a]	Month	Tractor	Operative machine			Notes Production factors
			Mass–Power	Type & Size	Time ($h\,ha^{-1}$)	FC[b] ($kg\,ha^{-1}$)	
Organic fertilisation, slurry spreading	1	April	7080 kg–130 kW	Slurry tanker 20 m^3	2.60	44.5[c]	85 t ha^{-1} pig slurry 0.24% N, 0.25 P_2O5, 0.55 K_2O
Organic fertiliser incorporation and primary soil tillage, ploughing	1	April	5050 kg–90 kW	3 ploughshares mouldboard plough depth 35 cm	1.66	24.9	
Secondary soil tillage, harrowing	2	April	4000 kg–73.5 kW	Rotary harrow	1.00	20.2	
Sowing	1	April	4900 kg–62.5 kW	Pneumatic precision seeder	0.50	8.4	19 kg ha^{-1} seed
Chemical weed control	2	April & May	4900 kg–62.5 kW	Pesticides sprayer	0.28	3.3	4 kg ha^{-1} lumax[d]; 1 + 1 kg ha^{-1} dual[e]; 60 kg ha^{-1} urea
Mineral fertilisation	1	May	4900 kg–62.5 kW	Centrifugal mineral fertiliser spreader	0.50	3.0	
Mechanical weed control	1	May	4900 kg–62.5 kW	Mechanical hoeing	0.83	4.2	
Irrigation	3	June/August	4900 kg–62.5 kW	Pump	1.10	12.6	1100 m^3 ha^{-1}
Harvesting	1	Sept.	8100 kg–110.3 kW	Combine harvester	2.00	42.0	
Transport		Sept.	5050 kg–90 kW	2 Farm trailers	2.00	15.1	
Grain drying	1	Sept.	4000 kg–73.5 kW	Dryer[f]	–	191[g]	
Straw management	1	Sept.	5050 kg–90 kW	Straw chopper	1.00	18.5	

[a] Repetitions.
[b] Fuel consumption.
[c] Fuel consumption = 55.6 kg ha^{-1} in AS2.
[d] Mesotrione 3.39% (37.5 g l^{-1}); S-metolachlor 28.23% (312.5 g l^{-1}); terbuthylazine 16.94% (187.5 g l^{-1}).
[e] S-metolachlor 86.5% (960 g l^{-1}).
[f] For 16 t of grain maize with 23% of moisture content.
[g] dm^3 of LPG.

Table 4
Parameters for N compounds emissions into the environment.

Parameter	Value
Characteristics of the organic fertiliser	Pig slurry[a]: dry matter content 1.89%, pH 7.5, total N content 2.43 kg t^{-1}, ammonia content 0.75 kg t^{-1}, P content 2.1 kg t^{-1} Digestate[b]: dry matter content 2.5%, pH 7.9, total N content 2.93 kg t^{-1}, ammonia content 2.47 kg t^{-1}, P content 2.4 kg t^{-1}
Air temperature during spreading	17 °C
Soil texture	Medium texture
pH	6.8
Cation Exchange Capacity	15.5 meq/100 g
Rainfall after the spreading	No rain in the first 3 days
Average wind speed	2 m s^{-1}
Atmospheric deposition of N	45 kg ha^{-1}[c]
Infiltration rate	High for pig slurry, medium for digestate
Maximum crop rooting depth	1.5 m
N content in maize grain	1.5% of dry matter[d]
Rainfall in winter season	350 mm
NH_3 emission factor for mineral fertilisers[e]	Urea: 15% of total applied mineral N CAN: 15% of total applied mineral N

[a] Lijó et al. (2014a,b, 2015).
[b] Bacenetti et al. (2014).
[c] EC, Joint Research Center.
[d] Idikut et al. (2009).
[e] Brentrup et al. (2000).

Table 5
Fertiliser related emission into air, soil and water.

Scenario	N-NH_3 ($kg\,ha^{-1}$)	N-N_2O ($kg\,ha^{-1}$)	N-NO_3^- ($kg\,ha^{-1}$)	N_2 ($kg\,ha^{-1}$)	PO_4^{---} ($kg\,ha^{-1}$)
BS	36.86	2.47	50.03	17.77	5.03
AS1	11.13	2.79	72.93	20.07	5.03
AS2	8.42	2.82	75.39	20.32	5.03
AS3	49.61	3.34	7.52	24.04	6.54
AS4	30.17	2.03	18.81	14.64	3.99
AS5	34.40	2.44	48.26	17.60	0.82
AS6	4.60	2.82	75.11	20.29	0.82

Table 6
Allocation factor for grain and straw in AS3.

Product	Yield (t ha^{-1})	Price (D ha^{-1})	Allocation factor (%)
Grain	12.3	180	82
Straw	14.5	35	18

2.6. 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) grain yield: minimum and maximum values recorded in the same farm during the previous 5 years were considered, first assuming the minimum (9.75 t ha^{-1} at commercial moisture) and then the maximum (14.50 t · ha^{-1} at commercial moisture) yield;
ii) maximum ammonia volatilisation losses for organic fertilisers: a variation of ±20% respect to the value assessed by the model EFE-So (55%) was considered.

The sensitivity analysis was performed on the BS scenario.

2.7. Life Cycle Impact Assessment (LCIA)

The environmental impacts have been estimated using the composite method recommended by the International Reference Life Cycle Data System (ILCD) (Wolf et al., 2012). The following impact

Table 7
Environmental impact for 1 t of maize grain at commercial moisture (14%).

Impact category	Unit	BS
Climate Change (CC)	kg CO_2 eq	2.36×10^{2}
Ozone depletion (OD)	kg CFC-11 eq	2.16×10^{-5}
Human toxicity with carcinogenic effect (HTc)	CTUh	7.99×10^{-6}
human toxicity with no carcinogenic effect (HTnc)	CTUh	1.02×10^{-4}
Particulate matter (PM)	kg PM2.5 eq	3.41×10^{-1}
Photochemical ozone formation (POF)	kg NMVOC eq	1.36×10^{0}
Terrestrial acidification (TA)	molc H+ eq	1.25×10^{1}
Terrestrial eutrophication (TE)	molc N eq	5.58×10^{1}
Freshwater eutrophication (FE)	kg P eq	1.74×10^{-1}
Marine eutrophication (ME)	kg N eq	2.86×10^{0}
Freshwater ecotoxicity (FEx)	CTUe	97.49×10^{2}
Mineral and fossil resource depletion (MFRD)	g Sb eq	2.97×10^{0}

categories were considered: climate change (CC; kg CO_2 eq), ozone depletion (OD; kg CFC-11 eq), particulate matter (PM; kg PM2.5 eq), human toxicity with carcinogenic effect (HTc; CTUh), human toxicity with no carcinogenic effect (HTnc; CTUh), photochemical ozone formation (POF; kg NMVOC eq), terrestrial acidification (TA; molc H+ eq), terrestrial eutrophication (TE; molc N eq), freshwater eutrophication (FE; kg P eq), marine eutrophication (ME; kg N eq), freshwater ecotoxicity (FEx; CTUe), and mineral and fossil resource depletion (MFRD; kg Sb eq).

3. Results

3.1. Baseline scenario

Fig. 3 shows the environmental hotspots for BS while Table 7 reports the absolute impacts for the FU.

In more details, the emissions caused by the application of the organic fertiliser in BS are responsible for more than 90% of TA, TE and ME and for 98% of FE. For the fertiliser emissions, the impacts are more relevant on PM and CC (78% and 42%, respectively). This is due to runoff, leaching and emissions to air caused by nitrate, ammonia, dinitrogen monoxide and phosphorous compounds. Slurry spreading affects HTnc for 22% (5.02×10^{-7} CTUh) and, although with a less extent POF (17%). Soil tillage (ploughing and harrowing) is one of the main contributors of POF (18%) and it cannot be neglected for OD, HTnc and MFRD (about 10%). Crop management (weed control, irrigation and top fertilisation) is the main hotspot for HTnc, HTc and MFRD (37%, 54% and 50%, respectively) while the harvest (harvesting and transport to the farm) for POF (28%). The main reason for OD, HTc, HTnc, POF and MFRD is the high fossil fuel consumption and related engine tractor emissions. Machinery wear (i.e. amount of machinery consumed) is important for HTc and MFRD. Drying plays a prominent role on OD (26%, mainly due to fossil fuel combustion for heat production) and CC (15%, exhaust gas emissions of dryer). Finally, pesticides emissions are almost completely responsible for the impact on FEx (98%), due to their harmful effect on freshwater, while their role on HTnc and HTc is negligible. Sowing and the production of seed, urea and pesticides do not represent hotspots for any impact category (<4% for all impact categories).

3.1.1. Sensitivity analysis results

The results of the sensitivity analysis are reported in Table 8.

Yield variability deeply affects the environmental results. In particular, yield reduction involves not only an impact increase due to the lower output from the system, but also due to higher emissions of N compounds. At lower yields is gathered a lower N removal from the soil and, consequently, higher nitrate leaching. ME, among the assessed impact categories, is the most affected by yield variation because of its close relationship with nitrate emission into water.

The variation of the maximum ammonia volatilisation losses has a little impact on the environmental performances of grain maize production (<1%) except for:

- PM (+12% and −10%, when maximum ammonia volatilisation losses increase and decrease of 20%, respectively),
- TA (+14.8% and −12.3%),
- TE (+14.9% and −12.4%),
- ME (+3.4% and −2.8%).

This confirms the relevance of ammonia emissions in air as important source of acidification and eutrophication.

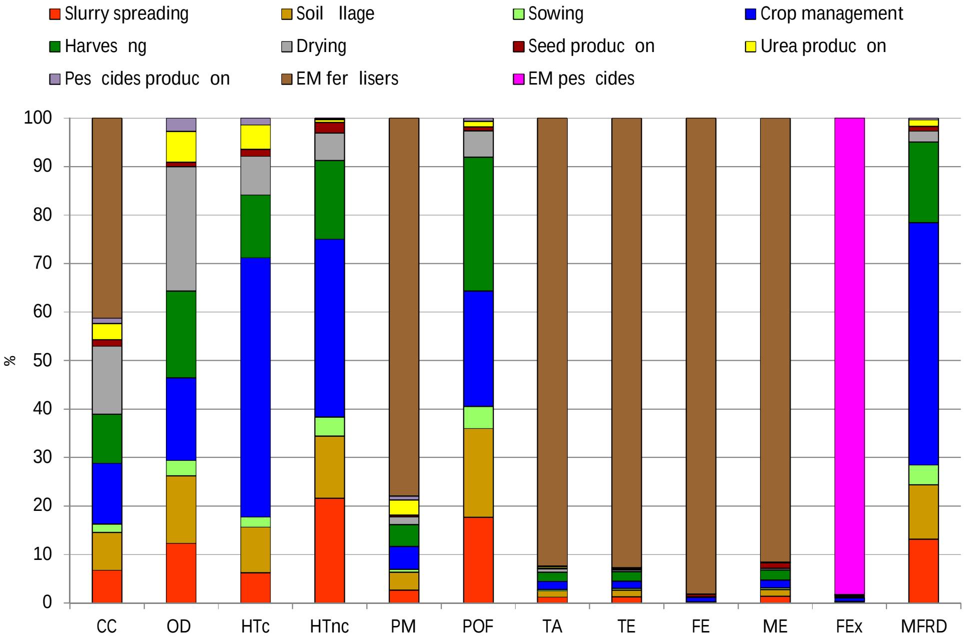


Fig. 3. Hotspots identification for BS.

Table 8
Results of the sensitivity analysis in which both NH_3 volatilisation and grain maize yield have been varied one by one. The results are given on all impact categories.

Impact category	NH_3 volatilisation for organic fert.		Grain yield	
	+20%	−20%	Max	Min
Climate Change (CC)	−1.0%	0.8%	−12.7%	21.2%
Ozone depletion (OD)	0.0%	0.0%	−11.3%	18.8%
Human toxicity with carcinogenic effect (HTc)	0.0%	0.0%	−14.0%	23.4%
human toxicity with no carcinogenic effect (HTnc)	0.0%	0.0%	−14.3%	24.0%
Particulate matter (PM)	12.0%	−10.0%	−14.9%	25.0%
Photochemical ozone formation (POF)	0.0%	0.0%	−14.3%	24.1%
Terrestrial acidification (TA)	14.8%	−12.3%	−15.0%	25.3%
Terrestrial eutrophication (TE)	14.9%	−12.4%	−15.1%	25.4%
Freshwater eutrophication (FE)	0.0%	0.0%	−15.2%	25.5%
Marine eutrophication (ME)	−3.4%	2.8%	−45.0%	75.7%
Freshwater ecotoxicity (FEx)	0.0%	0.0%	−15.1%	25.5%
Mineral and fossil resource depletion (MFRD)	0.0%	0.0%	−14.8%	24.9%

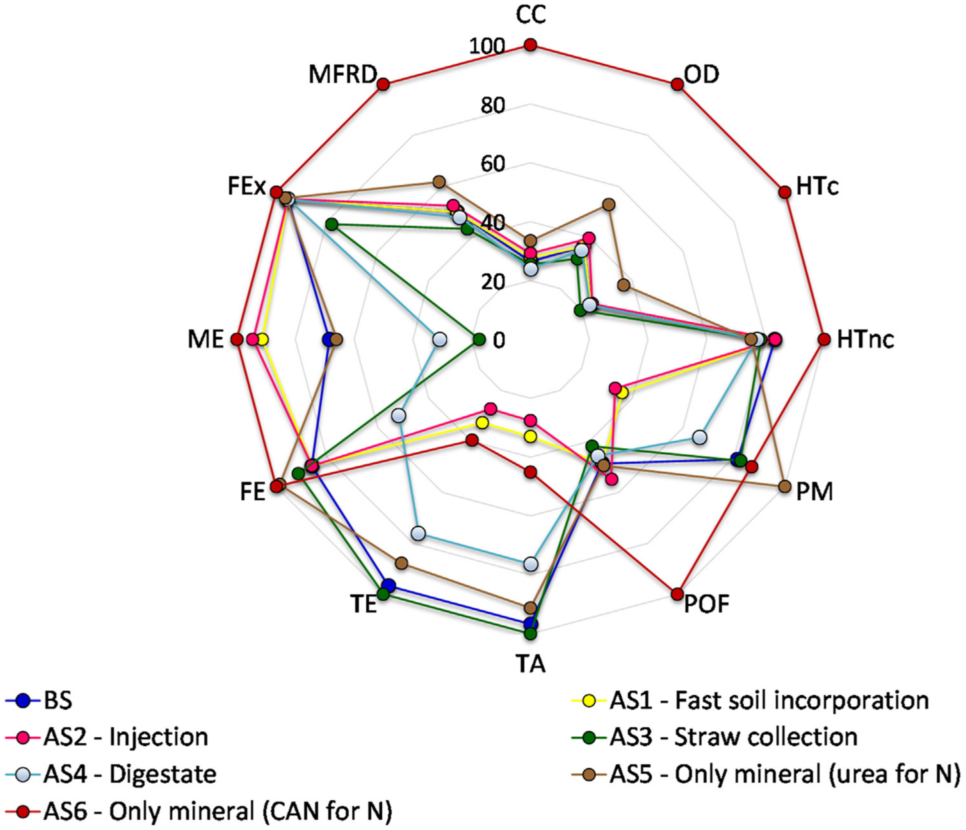


Fig. 4. Relative comparison of the environmental impact of the different scenarios (worst performing scenario = 100%).

3.2. *Comparison among the different scenarios*

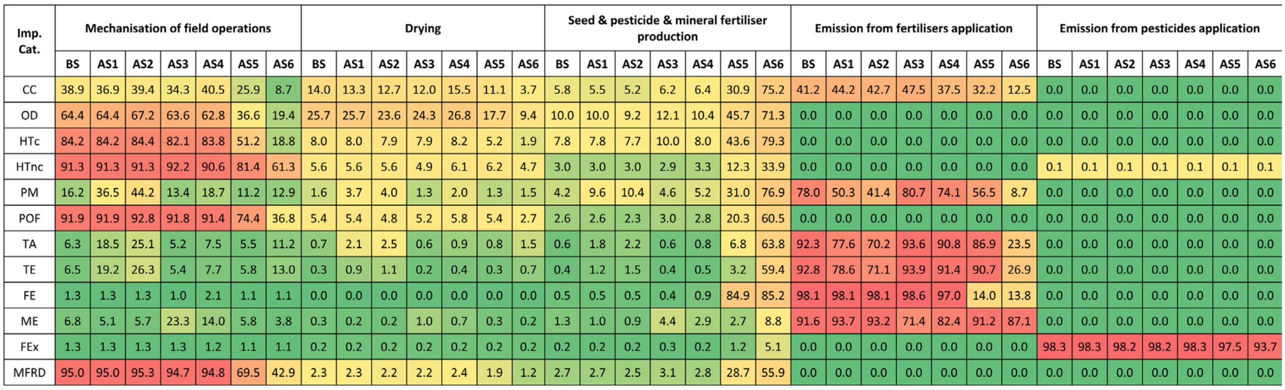
The relative comparison among these and BS is shown in Fig. 4. The results are reported as a percentage of the worst performing scenario. They are not univocal: there is not a best scenario that outperforms the others in all the evaluated impact categories.

Overall, AS6 is the worst scenario for 9 of the 12 evaluated impact categories. Respect to all the other scenarios wide differences emerge; in more details, the closest scenario for CC (−67%), OD (−52%), HTc (−63%), and MFRD (−39%) is AS5 while for HTnc (−15%) and POF (−45%) is AS2. When the AS6 is not considered, the differences among the other scenarios are reduced but AS5, the other scenario where only mineral fertilisers are spread, is the second worst scenario for 7 of the 12 evaluated impact categories.

AS3 is the worst scenario for 2 of the 12 evaluated impact categories: TA and TE, while for PM the highest environmental impact is related to AS5. For CC, OD and HTc a similar trend can be identified: AS6 and AS5 are by far the worst scenarios, AS3 and AS4 the best while BS, AS1 and AS2 show similar results, although AS2 has a slightly higher environmental impact. AS6 and AS5, where only mineral fertilisers (CAN-triple superphosphate and urea-triple superphosphate, respectively) are applied, show the highest impact, mainly due to the application of N fertiliser whose production is an energy-intensive process. Between these two scenarios, for PM, TA and TE the difference is related to the lower NH_3 emissions associated to CAN application instead of urea. For what concerns CC, the environmental difference among AS2 and the scenarios BS and AS1 is related to the higher fuel consumption for slurry spreading; in fact the injection requires higher traction force (due to the 5 anchors working at 7–8 cm depth) and, consequently, higher diesel requirement.

For HTnc, BS, AS1 and AS2 show a higher impact respect to AS3, AS4 and AS5, this latter has the lowest environmental impact (−14.7% AS6).

Table 9
Hotspots identification for each of the 7 scenarios distinguished per impact category. The values are percentages of the total impact.



Imp. Cat.	Mechanisation of field operations							Drying							Seed & pesticide & mineral fertiliser production							Emission from fertilisers application							Emission from pesticides application						
	BS	AS1	AS2	AS3	AS4	AS5	AS6	BS	AS1	AS2	AS3	AS4	AS5	AS6	BS	AS1	AS2	AS3	AS4	AS5	AS6	BS	AS1	AS2	AS3	AS4	AS5	AS6	BS	AS1	AS2	AS3	AS4	AS5	AS6
CC	38.9	36.9	39.4	34.3	40.5	25.9	8.7	14.0	13.3	12.7	12.0	15.5	11.1	3.7	5.8	5.5	5.2	6.2	6.4	30.9	75.2	41.2	44.2	42.7	47.5	37.5	32.2	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
OD	64.4	64.4	67.2	63.6	62.8	36.6	19.4	25.7	25.7	23.6	24.3	26.8	17.7	9.4	10.0	10.0	9.2	12.1	10.4	45.7	71.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HTc	84.2	84.2	84.4	82.1	83.8	51.2	18.8	8.0	8.0	7.9	7.9	8.2	5.2	1.9	7.8	7.8	7.7	10.0	8.0	43.6	79.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HTnc	91.3	91.3	91.3	92.2	90.6	81.4	61.3	5.6	5.6	5.6	4.9	6.1	6.2	4.7	3.0	3.0	3.0	2.9	3.3	12.3	33.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.1
PM	16.2	36.5	44.2	13.4	18.7	11.2	12.9	1.6	3.7	4.0	1.3	2.0	1.3	1.5	4.2	9.6	10.4	4.6	5.2	31.0	76.9	78.0	50.3	41.4	80.7	74.1	56.5	8.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
POF	91.9	91.9	92.8	91.8	91.4	74.4	36.8	5.4	5.4	4.8	5.2	5.8	5.4	2.7	2.6	2.6	2.3	3.0	2.8	20.3	60.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TA	6.3	18.5	25.1	5.2	7.5	5.5	11.2	0.7	2.1	2.5	0.6	0.9	0.8	1.5	0.6	1.8	2.2	0.6	0.8	6.8	63.8	92.3	77.6	70.2	93.6	90.8	86.9	23.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TE	6.5	19.2	26.3	5.4	7.7	5.8	13.0	0.3	0.9	1.1	0.2	0.4	0.3	0.7	0.4	1.2	1.5	0.4	0.5	3.2	59.4	92.8	78.6	71.1	93.9	91.4	90.7	26.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FE	1.3	1.3	1.3	1.0	2.1	1.1	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.5	0.5	0.4	0.9	84.9	85.2	98.1	98.1	98.1	98.6	97.0	14.0	13.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ME	6.8	5.1	5.7	23.3	14.0	5.8	3.8	0.3	0.2	0.2	1.0	0.7	0.3	0.2	1.3	1.0	0.9	4.4	2.9	2.7	8.8	91.6	93.7	93.2	71.4	82.4	91.2	87.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FEx	1.3	1.3	1.3	1.3	1.2	1.1	1.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.2	1.2	5.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	98.3	98.3	98.2	98.2	98.3	97.5	93.7
MFRD	95.0	95.0	95.3	94.7	94.8	69.5	42.9	2.3	2.3	2.2	2.2	2.4	1.9	1.2	2.7	2.7	2.5	3.1	2.8	28.7	55.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Note: CC = climate change; OD = ozone depletion; HTc = Human Toxicity with carcinogenic effect; HTnc = Human Toxicity with no carcinogenic effect; PM = particulate matter; POF = photochemical oxidant formation; TA = terrestrial acidification; TE = terrestrial eutrophication; FE = freshwater eutrophication; ME = marine eutrophication; FEx = freshwater ecotoxicity; MFRD = mineral, fossil and renewable resources depletion.

For PM, mainly due to ammonia emissions and mineral fertilisers production, AS5 is the worst scenario followed by AS6, the difference among these two scenario where only mineral fertilisers are applied is related to the low volatilisation rate of CAN (15% of the applied N) respect to urea (15% of the applied N). Respect to AS5, BS and AS3 show an impact reduction of about −20%, while AS1 and AS2, thanks to the substantial reduction of ammonia emissions related to fast soil incorporation and injection of the slurry, do even better (−63.4% and −65.8%, respectively).

For POF, excluding AS6 that is by far the scenario with the highest impact, AS2 shows the highest environmental impact, while the best performance is achieved by AS3 (−58.1%) and AS4 (−54.7%). BS, AS1 and AS5 achieve lower impact reductions (about −52% respect to AS2). The difference among the scenarios in which organic fertilisers are applied are related to diesel fuel consumption.

For TA and TE, AS3 is the worst scenario, while BS and AS5 lead to similar results (about −3%, −9% respectively). AS1 and AS2, thanks to ammonia emissions minimising due to different timing and organic fertiliser spreading technique, show by far the lowest environmental impact (respect to AS5, about −66% and −72%, respectively).

For FE, the scenarios with the worst performances are AS5 and AS6, mostly caused by the impact related to mineral fertilisers production and in particular to the manufacturing of triple superphosphate. Respect to AS5 and AS6, only AS4 achieves a significant impact reduction (−47.6%). For AS4, the higher MFE accounted for the digestate allows not only the spreading of a lower amount of organic fertiliser but also of a lower amount of P that, consequently, is less leached and lost by run-off.

For ME, AS6 is again the worst scenario but the difference with the solutions where volatilisation is reduced are little: AS2 (−5.6%), AS1 (−8.8%). Wider impact reductions are achieved by BS (−31.6%) and AS5 (−34.2%). AS4 and AS3 are the two scenarios with the lowest environmental impact (respect to AS6, −69.0% and −82.7%, respectively). For this impact category, the nitrate leaching plays a relevant role; in AS1 and AS2, ammonia emission reduction achieved thanks to the different timing and spreading technique of pig slurry involves higher nitrate leaching. AS3 is the best scenario not only due to allocation between grain and straw but also because straw removal reduces the nitrogen supply and, consequently, the N leaching during the winter season.

For FEx, AS3 is the only scenario where a considerable impact reduction (−21.6%) is achieved, this reduction is completely related to allocation.

For MFRD, AS6 shows a doubled impact respect to scenarios where organic fertilisers are spread, mainly because of fossil energy consumption during mineral fertilisers manufacturing. Among the other scenarios, AS5 is the worst and AS3 is the best (−38.2 and −56.7%, respectively). As for CC, the difference between BS-AS1 and AS2 is related to higher fuel consumption.

Table 9 reports the environmental hotspots for the different scenarios. The environmental impact related to the mechanisation of field operations such as the one related to production factors manufacturing (e.g., seed, pesticide and mineral fertilisers) have been gathered.

The hotspots analysis highlights similar trends among the different scenarios except for AS6 and AS5 that present the major differences respect to other scenarios. In more details:

- mechanisation of field operations is responsible for the main share of the environmental impact in OD (about 63–68%, except for AS5 where it is 37% and AS6 19%), HTc (about 82–84%, except for AS5 where it is 51% and AS6 19%), HTnc (about 82–92% except for AS6 where it is 61%), POF (about 74–93% but 37% for AS6) and MFRD (about 95%, except for AS5 where it is 70% and AS6 43%) and of about 26–40% of CC (except for AS6: 9%);
- grain drying, whose role is important only for CC (about 11–16%) and OD (about 9–27%). For both these two impact categories the share in AS5 and AS6 is slightly lower. This is not due to a lower absolute impact for drying (about 31.99 kg CO_2eq FU^{-1} for all scenarios, except for AS3 where allocation is carried out), but only to the higher absolute impact related to other processes (namely mineral fertilisers production);
- inputs production (seed, pesticides and mineral fertilisers), except for AS5 and AS6, is not an environmental hotspot in any scenario: their role is little (<10%) for CC, HTnc, POF, TA, TE, FE, ME, FEx and MFRD. On the contrary, in AS5 and AS6, inputs production is responsible for more than 10% of the total impact in 8 of the 12 evaluated impact categories and plays a prominent role in CC, OD, HTc, HTnc, PM, POF, FE and MFRD. This impact is mainly related to mineral fertilisers production and, in particular, to urea and CAN production and their related high energy consumption;
- the emissions related to fertiliser application have no impact in 6 of the 12 impact categories (OD, HTc, HTnc, POF, FEx and MFRD) but are relevant for PM, TA and TE that are affected by ammonia emissions, FE (about 97–99% for the scenarios 1–4, where losses of P compounds are high due to the high P amount applied with the organic fertiliser) and ME (about 71–94%, due to nitrate leaching);

- pesticide emissions are practically the only responsible for FEx (about 94–98% in all the scenarios).

4. Discussion

Other studies were previously carried out regarding the environmental impact of grain maize production (Bacenetti et al., 2015c; Bacenetti and Fusi, 2015; Goglio et al., 2012; González-García et al., 2012; Kim and Dale, 2008; Noya et al., 2015). Nevertheless, a direct comparison cannot be always drafted among the achieved results because of different system boundaries, functional unit, methodological assumptions (e.g., allocation) and LCIA method. For example, for CC impact category, which is not affected by the LCIA method used, the Ecoinvent database reports for Switzerland context a value of 610 kg of CO_2 eq t^{-1} and 438 kg of CO_2 eq t^{-1}, respectively for organic and integrated farming maize cultivation. These values are much lower than those found in this study. The differences with the impact assessed in this study are mainly related to lower grain yield (9.28 and 7.78 t ha^{-1} at commercial moisture for organic and integrated farming maize production, respectively) and to the mineral fertiliser consumption. In a study carried out by Noya et al. (2015) in the same geographic area (Po Valley), 5 different maize FAO classes were compared and, for CC, a value ranging from 346 to 586 kg of CO_2 eq t^{-1} was assessed. The CC value for FAO classes 600 and 700 (about 350 kg of CO_2 eq t^{-1}) is higher than the one achieved in this study for a maize hybrid with the same FAO class. In this case, the differences are not due to the grain yield, which is similar, but to fertilisation and irrigation. In Noya et al. (2015) both a higher amount of mineral fertiliser is applied in pre-seeding operation and a higher amount of diesel is consumed during surface irrigation carried out by pumps.

Regarding the different scenarios, the outcomes of this study show that with direct injection and – although with a lower extent – with fast incorporation into the soil, substantial environmental benefits can be achieved for those impact categories deeply affected by ammonia emissions (PM, TA and TE). Nevertheless, the downside cannot be neglected: without a correct quantification in space and time of the fertilisation, the minimisation of ammonia volatilisation could involve higher nitrogen leaching and, consequently, a remarkably higher impact for ME. Catch crops could be a suitable solution to reduce nitrate leaching and phosphorous run-off during the winter season and, consequently, to mitigate FE (affected by P losses) and ME (affected by nitrate). Nevertheless, it should be considered that the additional field operations as well as the seed consumption could affect the other impact categories.

For ME, which is mainly related to emissions of N compounds into water, small benefits respect to BS can be achieved when the fertilisation is carried out only with mineral fertilisation (AS5), using digestate instead of pig slurry (AS4) or collecting straw (AS3). Straw collection and sell involves a potential reduction of the environmental impacts as a consequence of allocation (about 18% of the total impact is allocated to straw). However, the additional organic fertiliser spreading required to compensate the higher nutrient removal and the related emissions offset this potential reduction and give rise to higher TA, FE and TE.

The use of only mineral fertilisers (AS5 and AS6) is not a valuable solution to reduce the environmental impact of maize grain production. The substitution of organic fertilisers with mineral ones does not involve environmental benefits for the impact categories affected by ammonia emissions arising from pig slurry spreading; on the opposite, it is related to impact increase for those impact categories affected by energy and fossil fuel consumptions (CC, OD, HTc and MFRD). Furthermore, the use of animal slurry as organic fertiliser responds to best agricultural practices and that whether not spread on fields, slurry management would be a notable issue, especially in areas devoted to livestock such as Northern Italy.

For what concerns the achieved results, it should be considered the following:

- the assessment is carried out considering that the different spreading techniques as well as the choice of one fertiliser instead of another does not affect the maize yield because the plant nutrient requirements are always satisfied. Whether yield was affected, the environmental results would be different (Amaducci et al., 2016);
- concerning the results for CC, due to the lack of experimental data, in the assessment no change in the soil organic matter content has been considered. For scenarios where organic fertilisers are spread, their application over a long period could involve an increase of carbon sequestration into the soil and, then, a reduction of CO_2 eq emissions. On the contrary, in AS3, the straw collection could involve a decrease of soil C sequestration and, therefore, a CC increase.

5. Conclusions

Organic fertiliser application is associated with emissions into air, soil and water that are responsible of not negligible environmental impacts. In this study, using the LCA method, the environmental impact of maize cultivation was evaluated taking into account fertilising schemes characterised by different organic and mineral fertilisers (pig slurry, digestate and urea, triple superphosphate and calcium ammonium nitrate) as well as different spreading techniques (direct soil injection and soil incorporation with different timing).

Although the achieved results are specifically referred to a case study and assessed considering that grain yield is not affected by changes in fertiliser type and spreading technique, they highlight the importance of organic fertilisers different application techniques. Depending on climatic and soil conditions as well as on physical and chemical fertiliser characteristics, by choosing the proper spreading technique (namely injection and fast soil incorporation), the environmental load can be steeply reduced for those impact categories affected by ammonia emission (e.g., particulate matter formation, terrestrial acidification, terrestrial and marine eutrophication). Nevertheless, this is a trade-off. If any of these solutions was implemented, a beneficial environmental effect would be not assured without taking into account that lower ammonia volatilisation involves higher nitrogen availability into the soil and that this could determine an increase of nitrogen leaching and of the related environmental impact (marine eutrophication). Consequently, a change in the amount of fertiliser spread is recommended to avoid the high leaching.

The achieved results are useful for the development of “spreading rules” able to drive the application of organic fertilisers in agricultural areas where there is an intense livestock activity. In fact, taking into account the environmental benefits and drawbacks that have arose from this assessment on the different type of fertiliser and organic spreading technique, can be useful for the development of targeted rules and policies.

Author contributions

All the authors conceived the work, JB collected and processed the data, JB and DL wrote and revised the manuscript.

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JB and DL wrote the paper and collected the inventory data; JB elaborated the inventory data; MF and JB revised the paper and planned the study.

Any opinions, findings, conclusions or recommendations expressed are those of the author(s) and do not necessarily reflect the views of the Departments involved in this study.

References

Althaus, H.J., Chudacoff, M., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. Life Cycle Inventories of Chemicals Ecoinvent Report No. 8, v2.0 EMPA. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Amaducci, S., Colauzzi, M., Battini, F., Fracasso, A., Perego, A., 2016. Effect of irrigation and nitrogen fertilization on the production of biogas from maize and sorghum in a water limited environment. Eur. J. Agron. 76, 54–65.

Amon, B., Kryvoruchko, V., Amon, T., Zechmeister-Boltenstern, S., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. 112, 153–162.

Bacenetti, J., Fusi, A., 2015. The environmental burdens of maize silage production: influence of different ensiling techniques. Anim. Feed Sci. Technol. 204, 88–98.

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., Fusi, A., Negri, M., Fiala, M., 2015a. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Bacenetti, J., Duca, D., Fusi, A., Negri, M., Fiala, M., 2015b. Mitigation strategies in the agro-food sector: the anaerobic digestion of tomato puree by-products. An Italian case study. Sci. Total Environ. 526, 88–97.

Bacenetti, J., Lovarelli, D., Ingrao, C., Tricase, C., Negri, M., Fiala, M., 2015c. Assessment of the influence of energy density and feedstock transport distance on the environmental performance of methane from maize silages. Bioresour. Technol. 466–467, 1066–1077.

Baldoni, R., Giardini, L., 2000. Coltivazioni erbacee Cereali e proteaginose. Ed. Patron, p. 410.

Bockstaller, C., Guichard, L., Keichinger, O., Girardin, P., Galan, M.B., Gaillard, G., 2009. Comparison of methods to assess the sustainability of agricultural systems. A review. Agron. Sustainable Dev. 29, 223–235.

Bodria, L., Pellizzi, G., Piccarolo, P., 2006. Il trattore e le macchine operatrici. Ed. Edagricole, p. 430.

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, 221–233.

Brentrup, F., Küsters, J., Lammela, J., Barraclough, P., Kuhlmann, H., 2004a. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertiliser use in winter wheat production systems. Eur. J. Agron. 20, 265–279.

Brentrup, F., Küsters, J., Kuhlmann, H., Lammel, J., 2004b. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology: I. Theoretical concept of a LCA method tailored to crop production. Eur. J. Agron. 20, 247–264.

Brockmann, D., Hanhoun, M., Négri, O., Hélias, A., 2014. Environmental assessment of nutrient recycling from biological pig slurry treatment—impact of fertilizer substitution and field emissions. Bioresour. Technol. 163, 270–279.

Carozzi, M., Ferrara, R.M., Rana, G., Acutis, M., 2013. Evaluation of mitigation strategies to reduce ammonia losses from slurry fertilisation on arable lands. Sci. Total Environ. 449, 126–133.

Ciampitti, I.A., Picone, L.I., Rubio, G., García, F.O., 2011. Pathways of phosphorous fraction dynamics in field crop rotations of the pampas of Argentina. Soil Sci. Soc. Am. J. 75, 918–926.

De Vries, J.W., Groenestein, C.M., Schröder, J.J., Hoogmoed, W.B., Sukkel, W., Groot Koerkamp, P.W.G., De Boer, I.J.M., 2015. Integrated manure management to reduce environmental impact: II. Environmental impact assessment of strategies. Agric. Syst. 138, 88–99.

Environdec, 2014. EPD Arable Crops., pp. 1–23, Available at: http://environdec.com/en/PCR/Detail/?Pcr=8804#.VaPW7F_tlBc.

FAOSTAT, 2016. Available at: http://faostat3.fao.org/home/E (accessed 05.02.16.).

Fedele, A., Mazzi, A., Niero, M., Zuliani, F., Scipioni, A., 2014. Can the Life Cycle Assessment methodology be adopted to support a single farm on its environmental impacts forecast evaluation between conventional and organic production? An Italian case study. J. Clean. Prod. 69, 49–59.

Ferrara, R.M., Carozzi, M., Di Tommasi, P., Nelson, D.D., Fratini, G., Bertolini, T., Magliulo, V., Acutis, M., Rana, G., 2016. Dynamics of ammonia volatilisation measured by eddy covariance during slurry spreading in north Italy. Agric. Ecosyst. Environ. 219, 1–13.

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

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Humbert, S., Köllner, T., Loerincik, Y., Margni, M., Nemecek, T., 2007. Implementation of Life Cycle Impact Assessment Methods Ecoinvent Report No. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Fusi, A., Bacenetti, J., 2014. Confronto tra modelli per la stima delle emissioni da fertilizzanti: il caso del mais da granella. In: Proceedings of 8th National Conference of Rete Italiana LCA, 19–20 June 2014, Florence, pp. 168–175.

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.

Gaudino, S., Goia, I., Grignani, C., Monaco, S., Sacco, D., 2014. Assessing agro-environmental performance of dairy farms innorthwest Italy based on aggregated results from indicators. J. Environ. Manage. 140, 120–134.

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

González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L., 2012. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J. Clean. Prod. 43, 71–83.

Halberg, N., van der Werf, H.M., Basset-Mens, C., Dalgaard, R., de Boer, I.J., 2005. Environmental assessment tools for the evaluation and improvement of European livestock production systems. Livest. Prod. Sci. 96, 33–50.

Hamelin, L., Naroznova, I., Wenzel, H., 2014. Environmental consequences of different carbon alternatives for increased manure-based biogas. Appl. Energy 114, 774–782.

Hasler, K., Bröring, S., Omta, S.W.F., Olfs, H.W., 2015. Life cycle assessment (LCA) of different fertilizer product types. Eur. J. Agron. 69, 41–51.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

ISO 14044, 2006. Environmental Management—Life Cycle Assessment—Requirements and Guidelines. International Organization for Standardization.

ISTAT, 2016. Available at: http://dati-censimentoagricoltura.istat.it/?lang=it (accessed 05.02.16.).

Idikut, L., Atalay, A.I., Kara, S.N., Kamalak, A., 2009. Effect of hybrid on starch, protein and yields of grain maize. J. Anim. Vet. Adv. 8 (10), 1945–1947.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J., 2007. Life Cycle Inventories of Bioenergy. Ecoinvent Report No. 17. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Kim, S., Dale, B.E., 2008. Effects of nitrogen fertilizer application on greenhouse gas emissions and economics of corn production. Environ. Sci. Technol. 42, 6028–6033.

Lazzari, M., Mazzetto, F., 2005. Prontuario di meccanica agraria e meccanizzazione. Ed. REDA.

Lebacq, T., Baret, P.V., Stilmant, D., 2013. Sustainability indicators for livestock farming. A review. Agron. Sustainable Dev. 33, 311–327.

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. Energy 68, 625–635.

Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Moreira, M.T., 2014b. Assuring the sustainable production of biogas from anaerobic mono-digestion. J. Clean. Prod. 72, 23–34.

Lijó, L., González-García, S., Bacenetti, J., Negri, M., Fiala, M., Feijoo, G., Moreira, M.T., 2015. Environmental assessment of farm-scaled anaerobic co-digestion for bioenergy production. Waste Manage. 41, 50–59.

Maguire, R.O., Sims, J.T., 2002. Soil testing to predict phosphorus leaching. J. Environ. Qual. 31, 1601–1609.

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

Minoli, S., Acutis, M., Carozzi, M., 2015. NH3 emissions from land application of manures and N-fertilisers: a review of the Italian literature. Ital. J. Agron. 3, 5–24.

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 Bioenergy 63, 321–329.

Negri, M., Bacenetti, J., Manfredini, A., Lovarelli, D., Maggiore, T.M., Fiala, M., Bocchi, S., 2014b. Evaluation of methane production from maize silage by harvest of different plant portions. Biomass Bioenergy 67, 339–346.

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Agricultural Production Systems. Final Report Ecoinvent v2.0 No. 15a. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, Switzerland.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C., 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Eur. J. Agron. 65, 40–51.

Niero, M., Ingvordsen, C.H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M.F., Hauschild, M.Z., Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds.), 2015. Springer, p. 415.

Noya, I., González-García, S., Bacenetti, J., Arroja, L., Moreira, M.T., 2015. Comparative environmental profile of cereal cultivation for feed and food production in Lombardy region (Italy). J. Clean. Prod. 99, 250–265.

Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds.), 2015. Case Studies, Methodological Issues and Best Practices. Springer, p. 415.

Prahsun, V., 2006. Erfassung der PO4-Austrage fur die Okobilanzierung SALCA Phosphor. Agroscope Reckenholz—Tanikon ART, pp. 1–20.

Regione Emilia-Romagna Regulation n. 161/2011, 2011. Provisions on the agronomic use of livestock effluents and water treatment arising from farms and small agro-food enterprises.

Solinas, S., Fazio, S., Seddaiu, G., Roggero, P.P., Deligios, P.A., Doro, L., Ledda, L., 2015. Environmental consequences of the conversion from traditional to energy cropping systems in a Mediterranean area. Eur. J. Agron. 70, 124–135.

Spielmann, M., Bauer, C., Dones, R., 2007. Transport Services. Ecoinvent Report No. 14. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Vu, T.K.V., Vu, D.Q., Jensen, L.S., Sommer, S.G., Bruun, S., 2015. Life Cycle Assessment of biogas production in small-scale household digesters in Vietnam. Asian Australas. J. Anim. 28, 716–729.

Wang, W., Dalal, R.C., 2015. Nitrogen management is the key for low-emission wheat production in Australia: a life cycle perspective. Eur. J. Agron. 66, 74–82.

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. International Reference Life Cycle Data System (ILCD) Handbook—Towards More Sustainable Production and Consumption for a Resource-efficient Europe. JRC Reference Report. European Commission—Joint Research Centre, Publications Office of the European Union, Luxembourg (EUR 24982 EN).

Wulf, S., Jäger, P., Döhler, H., 2006. Balancing of greenhouse gas emissions and economic efficiency for biogas-production through anaerobic co-fermentation of slurry with organic waste. Agric. Ecosyst. Environ. 112, 178–185.

Fuel and engine emissions during on-field tractor activity: a possible improving strategy for the environmental load of agricultural mechanisation

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Abstract

Agricultural machinery play an important role in the environmental sustainability assessments. In order to study with reliability the fuel consumption and the exhaust gases emissions from fuel combustion in the tractor engine, a case study was performed with field tests. During the trials several operations were monitored while carrying out the operations on field and the measured CAN-bus data and engine exhaust gases emissions (CO_2, CO and NO_X) were attributed to the field working states of effective work, turns at headlands and stops thanks to the GPS. Additionally, data during the farm-field transfers were also collected.

In addition to data processing from the field trials, a model for predicting fuel consumption and engine exhaust gases emissions was adopted and its reliability was studied for further future uses.

From the results, specific considerations about the tested tractor (Valtra N101, 82 kW maximum power, IIIA emissive stage) and the studied working conditions (e.g., engine speed, torque, working speed and depth) can be performed to get information valid for the engine and the operations. The final goal is to adopt trustworthy data on agricultural machinery operations for fulfilling inventories in Life Cycle Assessment (LCA) studies.

Keywords: Agricultural machinery; data processing; CAN-bus; exhaust gas analyser; efficiency improvement; environmental sustainability

1 Introduction

Thanks to the application of recent technology to agricultural machinery, and to tractors in particular, a great potentiality for the enhancement of efficiency and for the monitoring of engine variables has been proven (Pitla et al., 2016; Shadidi et al., 2014; Solaimuthu et al., 2015). Specifically, employing CAN-bus (Controller Area Network), data logging software, GPS (Global Positioning System) and exhaust gases emission analysers allows for collecting a huge amount of data, which are related to in-field activity, characterised by local reliability and geographical identification. To this, the growing interest in quantifying and reducing the environmental load of agricultural productions (Renzulli et al., 2015) must be added, which involves the possibility of adopting the abovementioned technology for a twofold scope: improving the machinery engineering and knowledge (Bishop et al., 2016) as well as the related environmental sustainability (Bacenetti et al., 2017).

For what concerns the environmental point of view, agricultural mechanisation is responsible for a substantial share of impacts (Niero et al., 2015; Silva Capaz et al., 2013), mainly due to fuel consumption and engine exhaust gases emissions and to the materials wear. The quantification of these impacts, at least for the mechanical field operations, still shows shortcomings (Lovarelli et al., 2017), but also room for improvement. In fact, collecting data and monitoring tractors' activity permits to improve the efficiency of tractors, their size and their use. This certainly presents advantages on the construction and management perspectives, but also on the environmental one (Lovarelli et al., 2016). Commonly, one of the most limiting factors to inventory data collection for the environmental impact assessment of agricultural machinery operations is the unfeasibility to collect or measure inventory data (i.e. primary source) because they can be time consuming, expensive and site and time dependent. Although primary data are the most reliable, the collecting difficulties and the limit of being site-specific cause the widespread use of secondary data (i.e. databases, scientific literature) that, on the other side, can be simplified and not fully reliable (Sala et al., 2017), especially if uncritically used (Bacenetti et al, 2017). Nevertheless, particularly for agricultural productions, the geographical (Perozzi et al., 2016), temporal and managerial characteristics (e.g., soil texture, field shape and slope, climate and seasonality, machinery adopted and management choices) deeply affect most environmental loads (Bacenetti et al., 2015; Lovarelli et al., 2017).

The availability on the market of tractors and implements equipped with new technology and of new techniques or management strategies increases the possibilities of collecting primary data (Marx et al., 2015). In particular, thanks to modern technology such as CAN-bus, GPS, electronic devices and exhaust gases analysers, a huge amount of data is accessible and measurable easily, constantly and simultaneously to the work on field (Fellmeth, 2003; Molari et al., 2013; Pitla et al., 2016). These data describe how the engine works, the fuel consumed and exhaust gases emitted and the working features and interactions in the tractor (Janulevičius et al., 2016), which encompasses the possibility of monitoring and mapping variables (Bietresato et al., 2015), of increasing the analyses reliability on modern machinery and of optimising inputs use and management (Larsson and Hansson, 2011; Lindgren and Hansson, 2002) due to the possibility of identifying the optimal combination of work conditions to reduce inputs use. In particular, manufacturers can use information about the effective usage on field of tractors to improve the construction and maintenance of tractors as well as to identify failures.

In this context, the possibility of performing field experiments to collect primary data is very promising both for reliable data collection and use and for the development of robust models for predicting the behaviour of engine-related variables. In particular, several prediction models exist in literature and one of them was adopted in this study; it uses engine specific coefficients that, although requiring detailed information for their quantification, ensure reliable assessments.

The general aim of the study is to make advances on the data and model availability related to the modern technology present on tractors, which results helpful for multiple scopes among which the improvement of the data reliability for sustainability evaluations completed by means of Life Cycle Assessment (LCA) method. The possibility of having trustworthy and specific data permits to calculate the environmental load of agricultural machinery operations in a reliable way, thus allowing to play a management role for the environmental sustainability and for introducing effective sustainability measures in the manufacturing field and in the farmers' perspective. For reaching this goal, the specific aims of this study are to:

(i) describe the experimental field trials carried out to collect primary data on mechanical field operations for cereal crops production, as well as the methodology that was adopted for the data processing and its possible future repeatability,

(ii) identify the most important data for the filling of reliable inventories of agricultural machinery field operations, thus showing what happens along the different working states of a single operation,

(iii) apply a reliable quantification model for the prevision of fuel consumption and exhaust gases emissions that takes into account the engine behaviour during the field operations,

(iv) show the discrepancies that can emerge in terms of description of field activities among measured data on field, data related to single working states respect to the whole field work as such and data from test benches, these last with regard mainly to engine exhaust emissions.

2 Materials and methods

2.1 Field trials

Data collection was performed directly during field trials in order to evaluate the real field working conditions and not the standardised bench testing ones.

The experiments were performed in Umeå (Sweden) at the Swedish Machinery Testing Institute and they involved performing field operations with the tractor Valtra N101, made available by the contractor company. **Table 1** reports the tractor characteristics.

Table 1. Tractor Valtra N101 characteristics.

Characteristic	Unit	Value
Rated power	P_{MAX}	82 kW
Rated engine speed	s	1860 rpm
maximum torque	M_{MAX}	490 Nm
Mass	m	4850 kg
Driving wheels	4 WD	
Emissive Stage	IIIA	
Engine emission abatement technology	EGR (Exhaust Gas Recirculation) for abating NO_X	

Valtra N101 was equipped with CAN-bus (Controller Area Network), GPS (Global Positioning System), Dewesoft® software for CAN-bus data collection and storage, and guidance control. Additionally, to measure the exhaust gases released during the field operations, Testo® 350 portable emissions gas analyser was used.

During trials, the following operations were monitored:

(i) ploughing, with a 3-furrow mouldboard plough,
(ii) harrowing, with a 3.0 m wide rotary harrow,
(iii) harrowing, with a 3.0 m wide spike harrow,
(iv) sowing, with a 6.0 m wide universal mechanical sowing machine,
(v) rolling, with a 5.4 m wide compactor roller.

2.2 Background and instrumentation used

Among the instrumentation developed to map, understand and study the activity of the tractor engine and of the related devices employable during on-field activity, the most widely used system is the CAN-bus (Controller Area Network). It is a serial high-speed wired data network connection that permits to electronic devices to communicate with each other and that, coupled with storing instrumentation, permits to collect huge amounts of data with high frequency (Speckmann and Jahns, 1999). CAN-bus is normed with SAE J1939 for the connections of electronic devices on agricultural machinery and with the standard protocol ISO 11898 (ISO, 2003).

It is commonly available on modern medium-high power tractors and has permitted to use and take advantage of electronics on agricultural machinery, in particular with the improvement in data monitoring and collection and in sustainability evaluations.

The data logger that was used for the acquisition and storage of CAN-bus data is Dewesoft® software that is equipped with the translation key from CAN-bus and uses more than 100 communication canals that can be selected. Already on-board it was possible to check how variables were changing over time, due to the interface available with an on-board-mounted personal computer that allowed to select the variables to show. The data collection and saving in Microsoft Office Excel format was performed for the subsequent processing phase.

The portable instrument for the measurement of engine exhaust gases is Testo® 350; it analyses the flux of gases from the exhaust pipe of the tractor and results the values in ppm (or in % for CO_2). The measured gases are NO_X, NO, NO_2, CO and O_2; CO_2 (%vol) is obtained from calculations deriving from O_2 concentration. In addition, the sample exhaust gas temperature (°C), the sample flow of exhaust gas (L min^{-1}; maintained as constant as possible by a pump) and the instrument temperature (°C) are also measured. Gas emissions (g h^{-1}) were calculated based on measured flow rates and concentrations with the methods described in Directive 97/68/EC.

It includes a stainless-steel gas sampling probe equipped with integrated thermocouples located close to the exhaust pipe. From here, gases reach Testo® 350 on-board of the tractor, equipped with up to 6 electrochemical (for NO_X – obtained as sum of NO and NO_2) and infrared (for CO) sensors, and it stores data in an on-board memory (up to 250,000 values). Digital sensors for calibration history and interference filter with electronic lifespan indicator are available as well as temperature monitoring and diagnostics are guaranteed by the instrument. The retention time ranges between 20 s and 40 s depending on exhaust gases. The instrument accuracy is high: for CO_2 is equal to ± 0.2% vol O_2; for CO ±5 ppm within a CO concentration value between 0-199 ppm and ± 5% mass for higher concentration (200-2000 ppm); for NO and NO_2, the accuracy is ±5 ppm within a NO and NO_2 concentration value between 0-99 ppm and ± 5% mass for a concentration of 100-2000 ppm and 100-500 ppm, respectively for NO and NO_2. A thermoelectric chiller removes moisture and every 30 minutes, for approximately 7 minutes, the analyser rinses from moisture the sensors and the analysis chamber. During this period, therefore, no emission measurement took place and the tractor was left on, in idling stationary conditions.

With the GPS (Global Positioning System), the position on field was identified to build a map in which the phases of working activity could be classified. The instrument's precision is characterised by less than 100 mm error. CAN-bus and the exhaust gases emission analyser detected engine and tractor data and, thanks to the GPS, all of them were attributed to a position on field.

2.3 Goal of the field trials

The aim of the field experiments is to collect data from CAN-bus and gases analyser in order to have information about the engine working features, fuel consumption and exhausts emissions while directly working on field in view of detailed and reliable Life Cycle Assessment (LCA) studies on agricultural machinery operations. LCA is a worldwide recognised method that permits to quantify the environmental impact of processes (ISO 14040 Series), for which inventory data concerning fuel consumption, engine exhaust gases emissions and the consumption of materials composing machinery represent essential information.

Thus, thanks to GPS were built maps of the field with CAN-bus and exhaust gases emissions variables, and these data were grouped in the following working states:

(i) effective work: condition in which the tractor is driving on the stretch effectively carrying out the operation;

(ii) turn at headland: condition in which the driver is manoeuvring at the headlands, including when the implement is lifted/lowered and/or turned before or after the turn;

(iii) stop: when the tractor is not moving, therefore its GPS position along time does not change. In this condition, often, the engine is idling, but this is not a compulsory condition;

(iv) transfer: the whole condition of transport from the farm to the field and vice versa.

To better study the role of the working states, the trials can be distinguished in two main parts:

(i) comparison of alternative headland strategies during an operation to study the behaviour of the engine within different conditions during the turns at the headlands;

(ii) completion of other field operations with defined engine and field working features to study the behaviour of the tractor in those conditions.

In both cases the aim is to identify the most relevant differences in terms of fuel consumed and exhaust gases released, what working conditions show the best outcomes on the environmental perspective and how can vary the fuel consumption and engine exhaust gases emissions by changing only few work conditions.

2.4 Description of the field trials

Several field operations were monitored during trials carried out in October 2016 on two sandy-loamy fields in collaboration with the Swedish Machinery Testing Institute (SMP) in Umeå (Sweden).

2.4.1 Alternative headland strategies

In the first field (area = 1.7 ha), rotary harrowing was carried out with the aim of comparing alternative strategies for completing the turns at the headlands. In more details, to perform these turns different driving schemes were used in accordance with practical farm working schemes and each of them was characterised by different engine running features. Five headland strategies were identified as shown in **Figure 1**; analysing all of them allowed to compare the engine use during the strategies and to identify the most beneficial on the environmental point of view and the improvable driving conditions that permit to reach lower fuel consumption and lower exhaust gases emission. Hence, the field was split in five areas; the dimension of each of them was defined in order to have a minimum number of turns (at least 10 for all operations) for repeatability in the statistical analysis.

Figure 1 around here

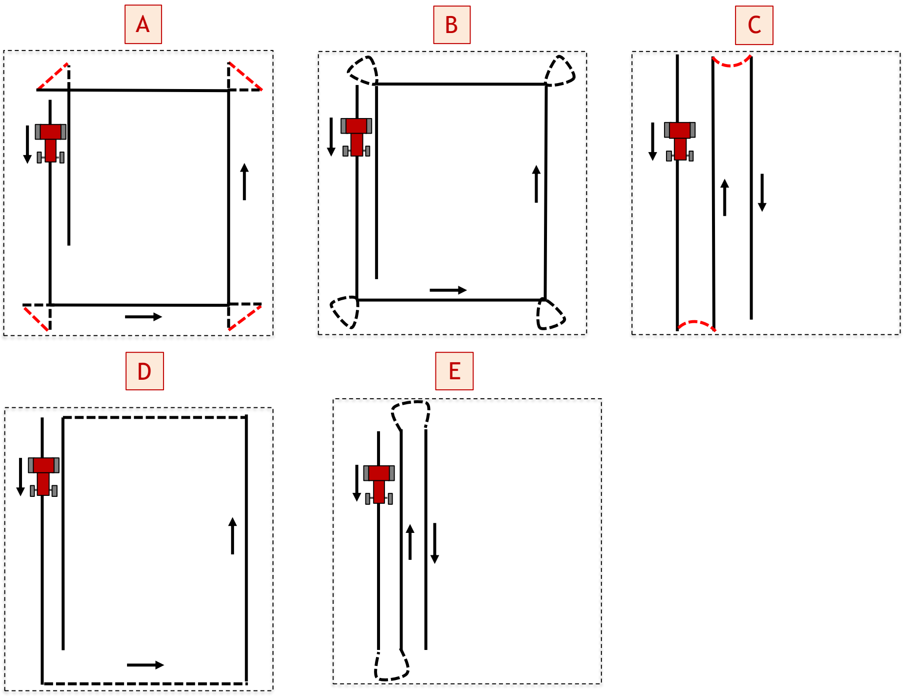

Figure 1. Studied headland strategies, namely A, B, C, D and E. The spotted lines identify the turn on the headlands, with the black-coloured line for the forward direction and the red-coloured line for the reversing.

In the sections of each area where the phase of effective work was carried out, the same working variables were considered, which means that gear, engine speed (rpm), working speed (km h^{-1}) and working depth (mm) were kept constant; the exception is the effective work on two areas, as reported in **Table 2**, where on 3 areas (i.e. I, II and V) the same engine speed and gear were kept during all the effective work, while in the remaining 2 areas (i.e. III and IV) engine speed or gear changed the way forward from the way back.

Table 2 around here

Table 2. Engine speed and gear used in the 5 areas characterised by different headland turning strategies.

Areas	Engine speed (rpm)	Gear (-)
I	$rpm_1 = 1850$	$g_1 = 2$
II	$rpm_1 = 1850$	$g_1 = 2$
III	$rpm_1 = 1700$ $rpm_2 = 2000$	$g_1 = 2$
IV	$rpm_1 = 1850$	$g_1 = 1$ $g_2 = 3$
V	$rpm_1 = 1850$	$g_1 = 2$

2.4.2 Other field operations

With regard to the operations of ploughing, spike harrowing, sowing and rolling, a second field characterised by an area = 4.2 ha was used (ploughing and rolling were performed only on one part of the field, with A_{plough} = 1.2 ha and A_{roller} = 2.8 ha). Similarly to rotary harrowing, data were collected during the work on field, taking into account the transfers from farm to field and vice versa and the work on field distinguished in effective work, turns at headlands and stops.

In each operation, engine speed and working speed were changed as reported in **Table 3**. When applicable (i.e. ploughing and spike harrowing) the working depth was also varied. The headland strategy was kept constant along the whole operation, but – when needed - differed in the different operations.

Table 3 around here

Table 3. Variables adopted in each operation.

Operation*	Headland strategy	Implement working width (b; m)	Implement working depth (H; mm)	Implement mass (m; kg)	Working speed (s; km h)	Tractor engine speed (n; rpm)
Ploughing (1-2)	D	1.47	H_1 = 180 H_2 = 280	1200 kg	s_1 = 5.0 s_2 = 7.0	n_1 = 1400 n_2 = 1800
Harrowing, rotary harrow (A-E)	A-B-C-D-E	3.0	100	890 kg	s_1 = 4.0 s_2 = 5.0 s_3 = 6.0	n_1 = 1700 n_2 = 1850 n_3 = 2000
Harrowing, spike harrow (1-4)	E	3.0	H_1 = 80 H_2 = 120	350 kg	s_1 = 6.0 s_2 = 8.0	n_1 = 1000 n_2 = 1400 n_3 = 1800
Sowing (1-2)	A-E	6.0	--	570 kg	s_1 = 5.0 s_2 = 8.0	n_1 = 1080 n_2 = 1800
Rolling	D	5.4	--	2460 kg	s_1 = 7.0 s_2 = 10.0	1000

* In brackets are shown the codes that identify the operations. More in details:

(i) ploughing 1 = work depth 180 mm; ploughing 2 = work depth 280 mm;

(ii) rotary harrowing A-E = A-E represent the 5 different headland strategies abovementioned;

(iii) spike harrowing 1 = all three engine speeds are studied one after the other on the same stretch; spike harrowing 2 = engine speed 1000 rpm; spike harrowing 3 = engine speed 1400 rpm; spike harrowing 4 = engine speed 1800 rpm;

(iv) sowing 1 = external part of the field with headland A; sowing 2 = internal part of the field with headland E.

2.5 Data processing of measured data

Collected data were processed on Microsoft Office Excel spreadsheet.

A first temporal offsetting of data from CAN-bus and Testo® 350 was made, and the identification of geographical coordinates from GPS followed.

As mentioned above, the working states were identified with the GPS coordinates considering that the worked fields had rectangular shape and, therefore: (i) when the GPS coordinates varied according to a defined angle the tractor was turning, (ii) when the GPS coordinates varied without exceeding the defined angle the tractor was working on the stretch (effective work) and (iii) when the GPS coordinates did not change for a period longer than 5 s, the tractor was stopping. An example of ploughing operation is shown in **Figure 2**.

Figure 2 around here

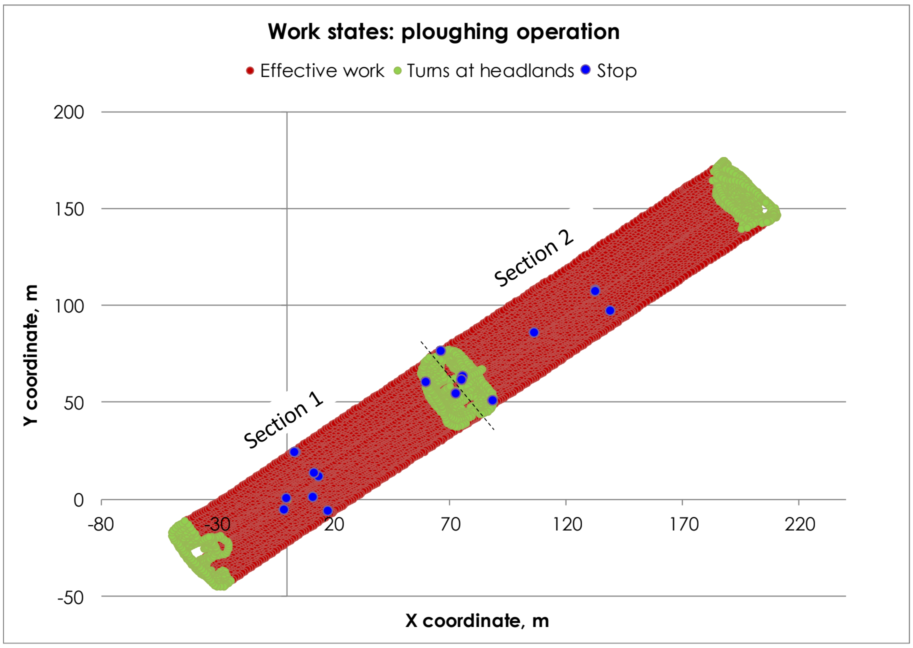

Figure 2. Distinction of work states for the ploughing operation. The working depth in Section 1 (left-bottom) was H_1 = 180 mm and in Section 2 (right-top) H_2 = 280 mm.

According to the working states distinction, the CAN-bus data related to torque (M; Nm), engine speed (s; rpm), fuel consumption (FC; L h^{-1}), engine power (P; kW), engine load (L; %), and the Testo® 350 data on exhaust gases emissions (EM of CO_2, CO and NO_X; g h^{-1}), O_2 (ppm) and instrument and gas temperatures (°C) could be attributed to each state.

In addition, in all operations, every stretch of effective work and every turn at headlands were numbered. This made possible to take mean values per stretch or turn and thereby quantify the stretch-to-stretch and turn-to-turn variation. Additionally, the specific values of brake specific fuel consumption (bsfc, g kWh^{-1}) and engine exhaust emissions (g kWh^{-1}) were also quantified in each stretch and turn, in order to be widely comparable among operations.

2.6 Predicting model adopted

From a literature analysis emerged that several prediction models for fuel consumption exist, some of which are based on generic equations (Grisso et al., 2004; Janulevičius et al., 2013; Sørensen et al., 2014) and others on engine-specific (Lindgren, 2004, 2005). Among them, the engine-specific one proposed by Lindgren (2005) has been adopted (Eq. 1). In this model, torque (M; Nm), engine speed (s; rpm) and engine-specific coefficients are needed. Torque and engine speed were directly gathered from the field measurements, while the 9 engine-specific coefficients were calculated referred to the equation (Eq. 1) modelling the semi-static condition (i.e. with no transient effect) and they were identified with Matlab® using a least square fit for calibration. Fuel consumption (FC; L h^{-1}) was quantified for all working states and total working time considering the following equation, which is also adopted for the quantification of EM (CO_2, CO and NO_X; g h^{-1}) with the related 9 engine-specific coefficients for EM. The total FC and EM for the operation is the sum of every value got per record of engine speed and torque.

$$FC = c_1 \cdot s + c_2 \cdot s^2 + c_3 \cdot s^3 + M \cdot (c_4 \cdot s + c_5 \cdot s^2 + c_6 \cdot s^3) + M^2 \cdot (c_7 \cdot s + c_8 \cdot s^2 + c_9 \cdot s^3) \qquad [1]$$

where:

- FC = fuel consumption (L h^{-1});
- from c_1 to c_9 = engine-specific coefficients;
- s = engine speed (rev min^{-1});
- M = torque (N m).

Data processing on engine exhaust gases is more complex because the production of each gas depends on a wide range of factors such as other gases present, temperatures, oxygen concentration, technologies and after-treatment systems and driving abilities (Larsson and Hansson, 2011; Lindgren and Hansson, 2002, 2004). However, Equation 1 responds well to engine exhaust emissions (Lindgren, 2005) and is valid for their quantification adopting adequate coefficients for each of the studied exhaust gases (see Table 7).

Lindgren (2005) studied two equations for fuel and exhaust emissions prediction: one assumes steady state conditions and one takes into account transient effects. Steady state is when no change occurs during the experiments for the measured data, whilst transient effects are changes due to fast variations in torque and/or engine speed. Transients are quantified evaluating the difference (%) from the steady state condition. As stated in Lindgren (2005), Equation (1) is valid for the steady state condition; the additional presence of three coefficients for the transients would permit to quantify FC and EM in transient conditions.

3 Results

Results are reported in two sections; first, on the processing of the measured data on field and then on the application of modelling.

3.1 Results on the measured data

For each operation, the working time was measured distinguishing in effective work, turns at the headlands, stops and the transfer from farm to field and vice versa. Results about the working time are reported in **Table 4** for all operations. In most cases, the effective work ranges between 60% and 70% of the total work time on the field (i.e. effective work, turns and stops without transfers), with a lower value for sowing (where stops are responsible for 29% of the total working time on field due to the filling of the hopper) and a higher value for rolling (which is a quite straight-forward operation). The turns at the headlands show a higher variability, ranging between 8% for rolling and 28% for rotary harrowing where the 5 headland strategies for the turns have been studied. For the stops, the result is affected by the rinsing of Testo® instrumentation that was performed with the tractor in a stationary idling position, as well as by the hopper filling during sowing. When considering the effective field work capacity, thus taking into account the transfers, the share of the total working time of the operation is affected by the distance from the field and influences the results; in particular, the contribution of transfers ranges between 17% and 56% for all the evaluated operations. Of course, considering the transfers (total working time of the operation = 100%), the work capacity on field decreases (i.e. effective work plus turns plus stops in a range between 44% and 83% of the total working time).

Table 4 around here

Table 4. Working time distribution (h) in the studied operations.

Operation	Effective work	Turns at headlands	Stops	Transfers	Total working time
Ploughing	1.93 h	0.62 h	0.39 h	1.46 h	4.40 h
Harrowing, rotary	1.77 h	0.82 h	0.36 h	1.40 h	4.35 h
Harrowing, spike	2.10 h	0.31 h	1.00 h [a]	0.70 h [b]	4.11 h
Sowing	0.69 h	0.16 h	0.35 h [c]	1.19 h	2.39 h

Rolling	0.29 h	0.03 h	0.09 h	0.53 h [b]	0.94 h

[a] This includes the time to couple tractor-implement on field (implement already on field) and to change the work layout of the implement (i.e. change of working depth between two field parts).

[b] The spike harrow and roller were already on the headlands of the field, therefore only the way back was measured. Thus, the total time (including way forward and way back) has been estimated.

[c] This includes the time to refill the hopper with seed.

The trend along time of the measured variables can be retrieved from the processing. **Figure 3** illustrates an example for this by focusing on two sections of rotary harrowing (i.e. headland turning strategies A and D) in which is also shown the distinction of collected data among effective work, turns at headlands and stops. In particular, when the headland strategy included changes in direction (e.g., strategy A), the trend in fuel consumption is widely variable (5-17 L h^{-1}), whereas when the turn is performed in a homogeneous driving scheme (e.g., strategy D) the fuel consumption is homogeneous and with a reduced variation level (5-8 L h^{-1} for most data). The variation in fuel consumption due to the effective work during the case of “headland strategy D” is consistent along the field in accordance with the change in gear (see Table 2, area IV).

Figure 3 around here

Figure 3. Trend along time of the measured fuel consumption for the rotary harrowing with the strategies for the headlands named “A” and “D”.

To get a value per stretch of effective work and per turn, every stretch and turn was numbered and for every of them it was possible to calculate statistics. In particular, **Figure 4** shows an example for ploughing, where every stretch and every turn are identified and report the average brake specific fuel consumption (bsfc; g kWh^{-1}) and CO_2 specific emission (EM_{CO2}; g kWh^{-1}). The specific values for bsfc and CO_2 were calculated considering the fuel consumed (L h^{-1}), CO_2 emitted (g h^{-1}) and absorbed engine power (kW) and averaging them per section of work state (i.e. per stretch and per turn). During the effective work, the values go up and down due to the field gradient that affected the tractor's developed engine power, which caused changes in brake specific fuel consumption and specific exhaust gases emissions between the way forward and the way back.

Figure 4 around here

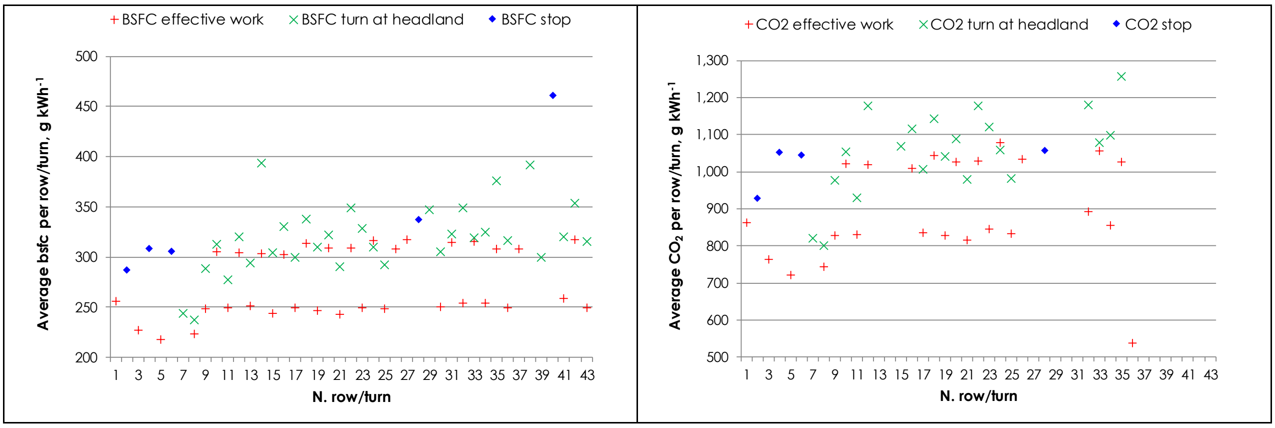

Figure 4. Average values for each work state of effective work, turn at headlands and stop for the ploughing operation (specific for Section 2 of ploughing). On the left, brake specific fuel consumption (g kWh^{-1}). On the right: specific values for CO_2 emission (g kWh^{-1}).

From the figure, it emerges that the specific values referred to the turns are higher respect to those during the effective work; thus, the efficiency of fuel (kWh g^{-1}) and the related one of CO_2 are better for the effective work state. It can also be seen that the stops play a role in regard of specific consumption and emission; in particular, although the stops are short in time, the bsfc and specific emission of CO_2 show values higher (340.3 g kWh^{-1} and 1020.7 g kWh^{-1} for bsfc and specific CO_2, respectively) than the average of turns (318.3 g kWh^{-1} and 1054.9 g kWh^{-1} for bsfc and specific CO_2, respectively) and, mainly, of effective work (274.2 g kWh^{-1} and 897.0 g kWh^{-1} for bsfc and specific CO_2, respectively), especially in the second part of the field.

Similarly, all results on the assessed operations that refer to the average bsfc (g kWh^{-1}), CO_2, NO_X and CO (g kWh^{-1}) per effective work, turn and stop are reported in **Figure 5**. Each operation was distinguished in different parts when different variables were considered (e.g., rotary harrow A-E for the 5 headland strategies, ploughing 1-2 for the two different working depths).

As expected, the specific values for fuel and exhaust gases emissions are almost always higher during turns at headland and stops rather than during the effective work on field due to the tougher working conditions, engine load and impact of transients. In particular, bsfc and CO_2 have a similar trend, due to their dependence on fuel use; instead, NO_X and CO show higher variability, mainly due to the EGR system, oxygen concentration and catalyst temperature.

Figure 5 around here

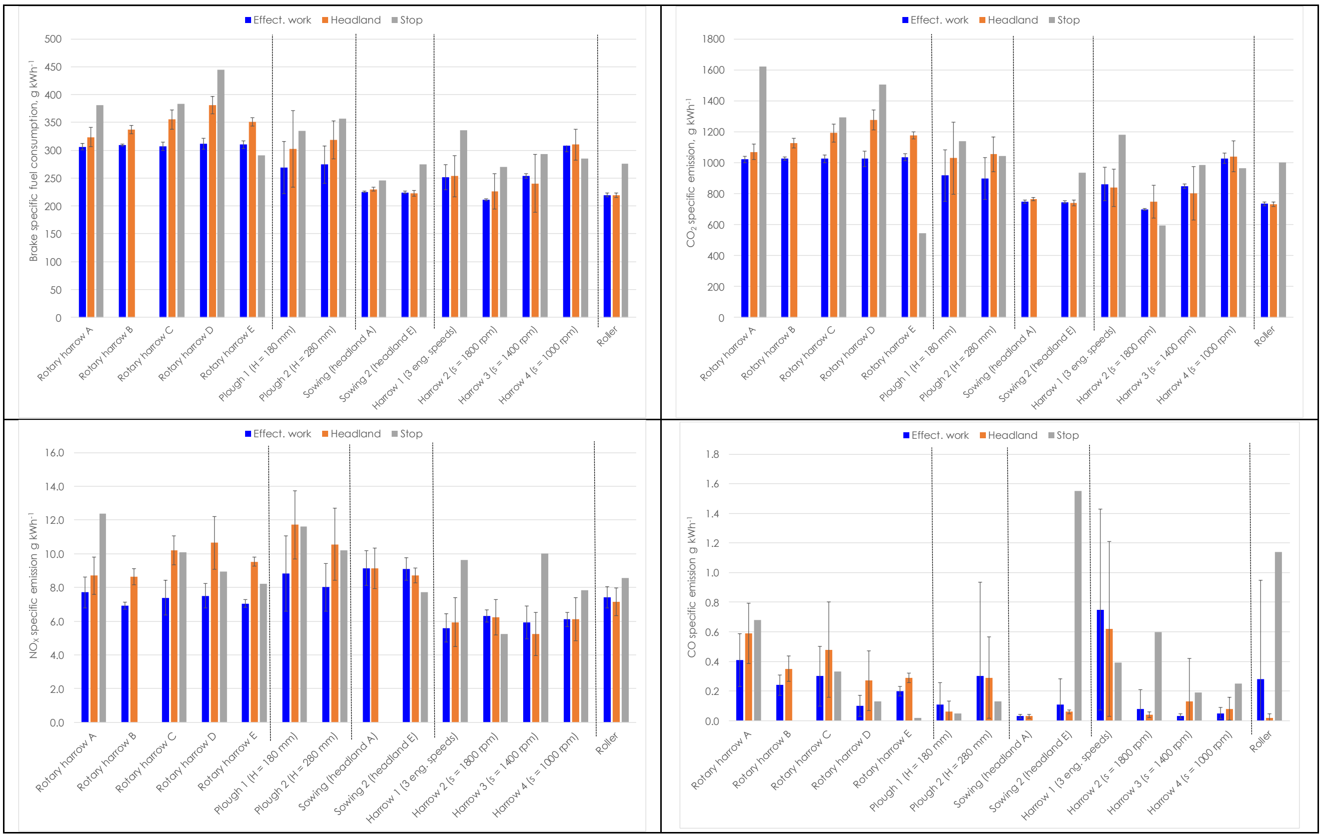

Table 4. Brake specific fuel consumption (bsfc; g kWh^{-1}), CO_2 (g kWh^{-1}), NO_X (g kWh^{-1}) and CO (g kWh^{-1}) average values per work state gathered from the data from field measurements. Standard

deviation is also reported for effective work and turns at headlands, while it was not calculated for stops due the low number of stops in the operation.

From the figure, a comparison among headland strategies can be performed within the rotary harrowing operation (i.e. headland strategies A-E). The results show that the highest values for bsfc, CO_2 and NO_X specific emissions are gathered during headland strategy D, followed by strategy C (-6.8% respect to turns in D) and strategy E (-7.9% respect to turns in D). For CO, instead, the opposite trend emerges, being strategy A the worst (followed by strategy C: -18% respect to A). Regarding the effective work, instead, the values are much closer to each other, as expected, due to the choice of adopting the same work conditions; however, higher values for bsfc and NO_X specific emission are shown in headland strategy D, where the turn strategy affected the effective work values as well. Another comparable operation is ploughing, where, however, not relevant differences emerge between the ploughing performed at 280 mm or at 180 mm depth (all values range within 89% and 100%, except for CO where lower results were highlighted during the first case).

The last comparable operation is the spike harrowing with options 2-4 (the variable is engine speed, with s_2 = 1000 rpm, s_3 = 1400 rpm and s_4 = 1800 rpm, respectively), from which it emerges that at lower engine speed the bsfc and the CO_2 specific emission were higher for all the three evaluated working states (other harrowing cases range for both variables between -16% and -32% of option 2); for NO_X as well as for CO, the best condition resulted the one in which the harrowing was performed at s_3 = 1400 rpm (effective work and turns at headlands) (-6% and -16% for NO_X during effective work and turns, respectively and even more for CO, for which, however, high variability is encountered) while the emissions during the stops were lower when the engine speed was s_4 = 1800 rpm (range between 42% and 85% respect to the worst case). In particular, the results obtained during the stops were affected by the fact that, when stops were shorter than 20 s, the engine speed was not idling but remained set at the work conditions.

For each variable is also reported the standard deviation of the operation and working state in order to understand how repeatable are the results. In most cases, standard deviation values are restrained, except for CO emission for which quite high values can be identified; moreover, in some operations such as ploughing 2 (H = 280 mm) and spike harrowing 1 (with the combination of 3 engine speeds one after the other on the same stretch; the field length was b = 420 m) show high standard deviations for torque and engine speed. Differences in these values can also be found from stretch to stretch and from turn to turn, mainly due to the specific field work conditions.

In order to understand in which working conditions, the Valtra N1010 engine performs the best in terms of bsfc and specific emission, the median values for bsfc, CO_2, NO_x and CO specific emissions (g kWh^{-1}) have been grouped according to engine speed and torque combinations (**Table 5**). In more details,

- engine speed is split in 3 groups: (A) s < 1100 rpm; (B) 1100 ≤ s < 1600 rpm; (C) s ≥ 1600 rpm;
- torque is split in 3 groups: (a) M < 100 Nm; (b) 100 ≤ M < 200 Nm; (c) M ≥ 200 Nm.

In this case, median was chosen since it resulted being a better indicator than average and mode.

Table 5 around here

Table 5. Median value of brake specific fuel consumption (g kWh^{-1}) and of specific emissions of CO_2, NO_x and CO (g kWh^{-1}) for each combination of engine speed and torque.

Variables	Combination of engine speed and torque								
	A-a	A-b	A-c	B-a	B-b	B-c	C-a	C-b	C-c
BSFC g kWh^{-1}	312.5	394.5	421.4	239.6	278.8	263.4	219.8	252.1	288.5
CO_2 g kWh^{-1}	907.9	1265.6	1338.2	760.1	884.3	1027.5	710.1	810.0	927.2
NO_x g kWh^{-1}	7.5	9.1	10.2	7.1	8.4	7.2	6.6	5.8	5.5
CO g kWh^{-1}	0.0	0.0	0.2	0.0	0.0	0.1	0.0	0.0	0.1

Notes: (A) s < 1100 rpm; (B) 1100 ≤ s < 1600 rpm; (C) s ≥ 1600 rpm; (a) M < 100 Nm; (b) 100 ≤ M < 200 Nm; (c) M ≥ 200 Nm.

In more details, the combined groups that include low engine speed values (i.e. A-a, A-b and A-c) are the less desirable solutions, since they show the highest values. High values mean that a worse efficiency is linked to this condition, characterised by the engine running slowly (idling or almost idling). The same trend is confirmed for CO_2 and NO_x. For nitrogen oxides, however, the trend involves also that at low torque (i.e. A-a, B-a and C-a) emissions are bigger than at high torque and, similarly to previous variables, they are the highest at low engine speed (s < 1100 rpm) followed by the intermediate step with medium-high torque and engine speed (B-b).

Regarding CO, the results are again more complicated to evaluate, although it emerges that CO specific emissions are higher with low engine speed (s < 1100 rpm) and high torque (≥ 200 Nm) (i.e. case A-c).

3.1.1 *Data processing of the transfer working phases*

The transfer phases were studied considering the complete transfer from farm to field and vice versa. This phase involves a wide range of variation in fuel consumption (1.5-24.9 L h^{-1}), due to the transferring on the paved road that involves fast travel speed changes.

In **Table 6** are reported the values of bsfc (g kWh^{-1}) and of the specific emission of exhaust gases (g kWh^{-1}) (when available) during each of the transfers studied for the field operations. Besides, also torque (Nm), engine speed (rpm) and engine power (kW) are given as average value of the transfer.

Table 6 around here

Table 6. Brake specific fuel consumption (bsfc, g kWh^{-1}), specific emission of CO_2, NO_X and CO (g kWh^{-1}), torque (Nm), engine speed (rpm) and power (kW) for the transfer phases. During part of the transfers, no information was collected on exhaust gases emission.

Work phases	bsfc g kWh^{-1}	CO_2 g kWh^{-1}	NO_X g kWh^{-1}	CO g kWh^{-1}	Torque Nm	Eng. speed rev min^{-1}	Eng. power kW
Transfer 1	371.79	0.32	27.50	4.17	134.0	1650.7	23.2
Transfer 2	412.53	--	--	--	93.49	1408.30	13.8
Transfer 3	429.86	0.58	51.55	0.64	63.06	1094.95	7.3
Transfer 4	452.18	0.00	0.02	0.00	84.0	1220.8	10.7
Transfer 5	268.01	--	--	--	116.5	1019.7	12.4

3.2 Results on the modelled data

The 9 coefficients needed for modelling fuel consumption and engine emissions for the engine of tractor Valtra N101, in accordance with the model described in Section 2.6, are shown in **Table 7**. For both fuel consumption and emissions, they were calibrated with the measured values.

Table 7 around here

Table 7. Model engine-specific coefficients calculated for tractor Valtra N101.

Engine-specific coefficients	Variable			
	Fuel consumption	CO_2 emission	CO emission	NO_x emission
C_1	$-2.29\cdot10^{-3}$	$-5.57\cdot10^{0}$	$4.33\cdot10^{-2}$	$-3.95\cdot10^{-1}$
C_2	$4.35\cdot10^{-6}$	$1.12\cdot10^{-2}$	$-6.77\cdot10^{-5}$	$6.27\cdot10^{-4}$
C_3	$-1.10\cdot10^{-9}$	$-2.90\cdot10^{-6}$	$2.67\cdot10^{-8}$	$-2.14\cdot10^{-7}$
C_4	$5.92\cdot10^{-5}$	$1.49\cdot10^{-1}$	$-1.52\cdot10^{-4}$	$5.74\cdot10^{-3}$
C_5	$-5.15\cdot10^{-8}$	$-1.26\cdot10^{-4}$	$2.80\cdot10^{-7}$	$-7.04\cdot10^{-6}$
C_6	$1.91\cdot10^{-11}$	$4.81\cdot10^{-8}$	$-1.46\cdot10^{-10}$	$2.38\cdot10^{-9}$
C_7	$-1.18\cdot10^{-7}$	$-3.04\cdot10^{-4}$	$2.66\cdot10^{-7}$	$-1.19\cdot10^{-5}$
C_8	$1.64\cdot10^{-10}$	$4.27\cdot10^{-7}$	$-5.97\cdot10^{-10}$	$1.64\cdot10^{-8}$
C_9	$-5.35\cdot10^{-14}$	$-1.42\cdot10^{-10}$	$4.31\cdot10^{-13}$	$-5.85\cdot10^{-12}$

As stated in Lindgren (2005), adopting the equation that evaluates transient effects permits to reduce the model error. Nevertheless, for these field experiments, the equation (Eq. 1) in steady state conditions was selected. The reason is related to the analysis performed on transients (i.e. the rate of change in engine speed per second over the maximum engine speed of the engine): their effect on all studied operations is reduces, as shown in **Figure 7**. The difference in the colours is related to the couple Engine Speed-Torque, which was made in order to identify the possible differences in transient due to the relation between engine speed and torque; the adopted couples "engine speed-torque" were built splitting engine speed in three groups ($s < 800$ rpm; $800 \leq s < 1200$ rpm; $s \geq 1200$ rpm) and torque in three groups ($M < 50$ Nm; $50 \leq M < 150$ Nm; $M \geq 150$ Nm) and matching the combinations. The values that constitute the grouping differ from the former ones because, in this case, it was more important to focus on the phases in which transients can play a prominent role, thus splitting with bigger detail the sections with low engine speed and torque. The graph is aimed to show the impact of the transient respect to fuel consumption modelling with the steady-state condition (Y-axis, left) at different transient presence, both negative and positive transients (X-axis). It can be seen with the triangular dots in the figure (Y-axis, right) that most data (96.0%) are enclosed in the range ±5% of transient effect; instead, in the range ±10% are included 99.3% of all data. Considering the range ±5%, the impact of the transient is very restrained, which explains why the steady state modelling equation was adopted.

Figure 7 around here

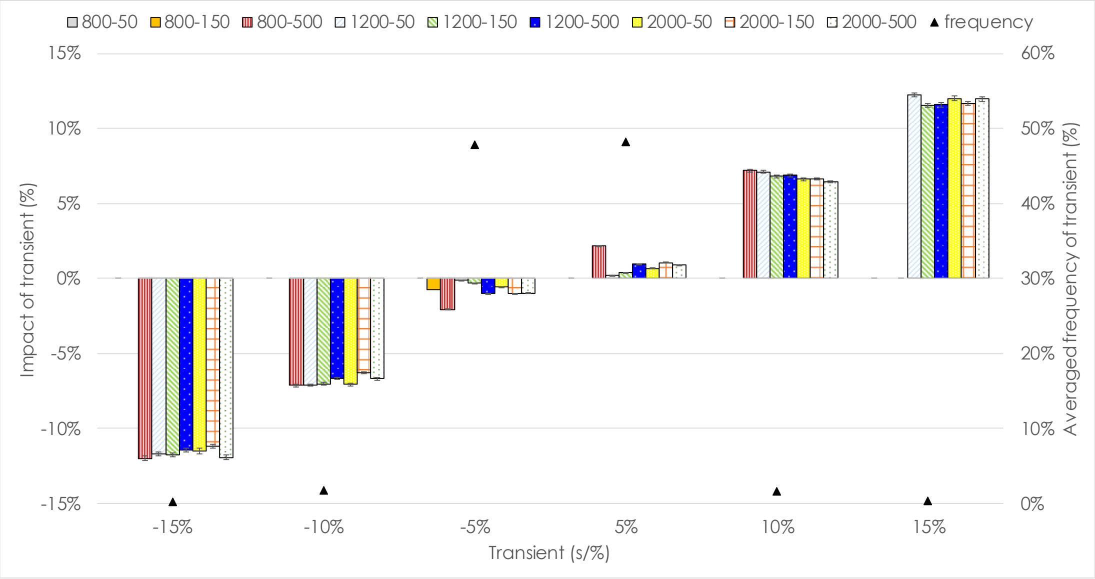

Figure 7. Transients effect during all the studied operations. The legend reports the combination of values of engine speed (s < 800 rpm; 800 ≤ s < 1200 rpm; s ≥ 1200 rpm) and torque (M <50 Nm; 50 ≤ M < 150 Nm; M ≥ 150 Nm) per series. The triangle-dots show the averaged frequency of transients.

The model described very well the engine, and mostly the fuel consumption and CO_2 emissions; for NO_x and, mainly, for CO more variability must be considered and therefore the model outcomes are less performing. Carbon monoxide (CO) was subject to excessive unrepresentativeness from the steady state model and usually depends on unregular conditions. In fact, CO is affected by several variables (Lindgren, 2005), among which air supply and the abilities of the driver, motivating the not sufficient response to the model. **Table 8** reports the R^2 that describe the model response to fuel consumption and exhaust gases emissions.

Table 8 around here

Table 8. Values of R^2 for the model used in predicting fuel consumption and engine exhaust gases emissions.

Work state	R^2			
	Fuel consumption	CO_2 emission	NO_x emission	CO emission
Effective work	0.97	0.90	0.22	0.19

Turns at headland	0.92	0.77	0.38	0.32
Stops	0.95	0.65	0.42	0.05

Nitrogen oxides (NO_X) depend on the internal engine temperature and the higher the temperature the higher is the exhaust gas emission. In this case, the tractor is equipped with the EGR, which involves that when the exhaust gases reach a defined threshold for temperature, the EGR system starts and brings to NO_X reduction. This works mostly during the effective work; instead, during turns the temperature in the exhaust pipe varies more in accordance with the working conditions, therefore higher variation can be identified. For what regards the stops, the engine is commonly idling and the EGR does not start working, at least until the threshold temperature is reached. Due to the lower temperature, NO_X emission values are lower. The main problems in this case are, however, that: first, some measured data reach very high values, probably due to the working conditions and sensibility of the instrument and, second, the temperature fast variation cannot be correctly identified with the model.

Considering the general model's response with all data processed and considering effective work, turns and stops, the model calculated values were, on average: (i) +4% respect to the measured bsfc (coefficient of variance = 0.09), (ii) -1% respect to the measured CO_2 (coefficient of variance = 0.39), (iii) +4% respect to the measured NO_X (coefficient of variance = 0.39), and (iv) +2% respect to the measured CO (coefficient of variance = 0.60). It can be observed that most differences are related to bsfc and NO_X, for which, however, the reasons are connected to the higher data availability for bsfc, since differently from emissions there is no rinsing; moreover, most discrepancies from the measured values are related to the turns and stops where the impact of the transients, although restrained, plays a more important role respect to the effective work phase. In support of this, if the model was not used for the stops, the outcomes would be included within ±2% respect to the measured values for all the 4 variables (bsfc, CO_2, NO_X and CO specific emission). The coefficient of variance for the measured and calculated values is close to 0 for most data, except for the section characterising the stops of two field operations (i.e. section of rotary harrowing and section of ploughing). The good response in this case is also motivated by the fact that, being an average for all data, variability is averaged along the whole dataset.

4 Discussion

In the current study are reported the results of field experiments carried out with one tractor coupled with several implements; in particular, the aim was first to measure all variables affecting the work of the tractor engine, the fuel consumption and the engine exhaust gases emissions, and second to use a model that could satisfactorily describe the system with the goal of having reliable data to adopt in Life Cycle Assessment studies.

From the results, it emerges that a high-level modelling can be reached by monitoring field operations through the electronic instrumentation, which is a very useful step forward to efficiency increase, inputs use and agricultural sustainability assessment. In particular, an interesting finding was the possibility of showing that working states highlight strong differences respect to each other and that studying what working state compose the operation is important.

Collected data on field only describe the specific tractor's engine tested that was built to match the IIIA Emissive Stage restrictions and, therefore, was equipped with the EGR system for the reduction of NO_X emissions. Older engines as well as newer ones that must respect the legislation with Stage IIIB (presence of Selective Catalytic Reduction - SCR - with Adblue) are likely to have a different dynamic. Thus, the results of this study are not applicable to other tractors/engines in their specific terms of the resulting values, but they are widely applicable in general terms when focusing on the engine's behaviour and on the methodology in building a model.

An additional plus is given by the fact that, usually, studies refer to test bench measurements and to the operating points defined by the ISO 8178-C1 Standard (ISO, 1996), whilst in this study the measurements were done directly on field, involving that higher variability due to the effective field work conditions should be taken into account, especially with regard to engine exhaust gases emissions (Larsson and Hansson, 2011; Lindvall et al., 2015). Having data directly measured on field makes values not comparable with other operations but permits to reliably describe the effective work conditions under assessment, without underestimates of variables due to the test bench. However, test bench measurements can be still efficaciously used to produce the coefficients for the steady state modelling, which permits to gather these coefficients without specifically performing tests on field and, thus, to fasten reliable data collection for subsequent environmental assessments.

Studying the different headland strategies was aimed to show whether and to what extent the operating modes on the headlands affect fuel and exhaust gases emissions (Janulevičius et al., 2013). More complex headland strategies involved, in fact, higher specific fuel consumption and higher specific engine emissions. In addition, the distinction of the field in the states of effective work, turns and stops permitted to understand if and how the fast variation in engine features such as engine speed and torque causes specific increases in consumption and emission. As expected, the specific values gathered during the stops involve an increase in brake specific fuel consumption and specific exhaust gases emissions, causing higher costs for fuel and higher environmental air pollution. The best efficiency of fuel is related to the effective work for almost all studied operations.

Considering data processing, the model for steady state conditions was adopted, since the studied operations highlighted a low impact of transients; this means that extending this model to transient effects was not expected to give important benefits on the modelling. However, the extension can be useful in predicting fuel and emissions (Lindgren, 2004) when transients are more present respect to these trials (e.g., during front loading operations).

The adopted model gave a very good response to fuel consumption and CO_2 emission. However, for NO_X and CO, it underestimated the real emission, probably due to the transient effects playing a greater role on these emissions rather than on fuel and CO_2. Moreover, CO is affected by air supply and incomplete combustion (Lindgren and Hansson, 2004). With positive transients, CO emissions increased because of the incomplete combustion, whereas during negative ones the emissions were close to the steady state condition. However, CO has resulted being subject to hikes and with an unregular trend also in other studies. Considering NO_X emissions, instead, commonly what occurs is that at higher temperatures the NO_X emissions increase (Janulevičius et al., 2013); in this case, on the contrary, when the threshold temperature was reached, the EGR system started working, therefore when temperatures increased considerably, the NO_X did not follow the trend. Consequently, with the EGR, NO_X emissions reduced (condition that usually occurs during the effective field work - medium-high torque and medium-high engine load - while increased during the accessory working time). Given this wide variability in the modes to reduce exhausts, a trade-off among them must be found.

For what concerns the transport phases, instead, the transient effects had higher importance than those on field and, in fact, the steady state model works less well. In particular, considering that farms are becoming fewer but bigger and that farmers need to drive longer distances to reach fields from farm, especially on an environmental perspective the transfer distances, engine features during transfers as well as fuel consumption and exhaust gases emissions of these accessory work phases are becoming increasingly important.

The results can be widely applicable, both to estimate variables that can be adopted in other models and to fill in the inventories for Life Cycle Assessment (LCA) studies in order to quantify appropriately the environmental impact of agricultural field operations (Larsson and Hansson, 2011; Bacenetti et al., 2017), providing more reliable results on specific studied cases. Different working conditions and implements (Lovarelli et al., 2017) as well as exhaust gases emissions from tractors equipped with different emission control strategies and/or engine emissive stages (Bacenetti et al., 2017) can be consistently evaluated, and adequate mitigation strategies for agricultural production chains can be identified (Renzulli et al., 2015). Considering the effect of fuel and exhaust gases, the environmental assessment through LCA is very important, since fuel consumption, CO_2 and NO_X are important sources of environmental impact. Fossil resources affect several environmental impact categories, such as Climate Change, Ozone Depletion, Terrestrial Acidification, Marine and Freshwater Eutrophication, and Mineral, Fossil and Renewable Resources Depletion (Wolf et al., 2013). On the other hand, CO plays an important role on human health, although it is commonly less important from an agricultural perspective due to the lower population density that lives in the countryside where most agricultural activities occur.

5 Conclusions

The study was aimed to report the results of measurements deriving from trials on field with different field operations and to apply a model that could describe the tractor’s fuel consumption and exhaust gases emissions with reliable results. Every data was related to a work state on field to show what occurs during each field work state within different work conditions (e.g., working speed, working depth, engine speed, engine load). This permitted also to make statistics on the most frequent work conditions and engine features that characterise agricultural machinery field operations. However, it is fundamental to underline that the results only refer to the tested engine. The use of such values in the completion of the inventory for environmental sustainability studies permits to improve the reliability of LCA results and, therefore, to make valid assessments that allow for suggesting valid environmental mitigation strategies. In more details, focusing on the effective working conditions on field permits: (i) to avoid underestimations or overestimations as due to bench tests, (ii) to quantify the difference between the most sustainable solution and the other alternatives for the farmer, (iii) to understand where improvements can be introduced along the work stages on field and, finally, (iv) to make farmers conscious of their role on the environmental sustainability of agricultural productions.

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Authors contributions

DL planned the field trials, DL and GL planned the study, processed the data and wrote the paper. All authors revised the paper.

References

Bacenetti, J., Fusi, A., Negri, M., Fiala M. (2015). Impact of cropping system and soil tillage on environmental performance of cereal silage productions. Journal of Cleaner Production, 86, 49-59.

Bacenetti, J., Lovarelli, D., Fiala, M. (2017). Environmental impact assessment of field mechanisation for a sustainable agriculture. 1st AXEMA-EurAgEng Conference. Villepinte 25 February 2017.

Bietresato, M., Calcante, A., Mazzetto, F. (2015). A neural network approach for indirectly estimating farm tractors engine performances. Fuel, 143, 144–154.

Bishop, J.D.K., Stettler, M.E.J., Molden, N., Boies, A.M. (2016). Engine maps of fuel use and emissions from transient driving cycles. Applied Energy, 183, 202-217.

European Directive 97/68/EC. On emissions from non-road mobile machinery. In: Official Journal of the European Communities 1998. L 59: 1.

Fellmeth, P. (2003). ISO11783 a Standardized Tractor – Implement Interface. iCC 2003 CAN in Automation, 8–13.

Grisso R.D., Kocher M. F., Vaughan D.H. (2004). Predicting tractor fuel consumption. Biological systems engineering: papers and publications. Paper 164.

ISO 14040 series, 2006. Environmental management – Life Cycle Assessment – Requirements and guidelines. International Organization for Standardization.

ISO 8178-4. (1996). Reciprocating internal combustion engines – exhaust emission measurements – part 4: Test Cycles for Different Engine Applications. International Organisation of Stadardisation. Geneva, Switzerland.

Janulevičius, A., Juostas, A., Čipliene, A. (2016). Estimation of carbon-oxide emissions of tractors during operation and correlation with the not-to-exceed zone. Biosystems Engineering, 147, 117-129.

Janulevičius, A., Juostas, A., Pupinis, G. (2013). Engine performance during tractor operational period. Energy Conversion and Management, 68, 11-19.

Larsson, G., Hansson, P-A. (2011). Environmental impact of catalytic converters and particle filters for agricultural tractors determined by life cycle assessment. Biosystems Engineering, 109, 15-21.

Lindgren, M., Hansson, P-A., 2002. Effect of engine control strategies and transmission characteristics on the exhaust gas emissions from and agricultural tractor. Biosystems Engineering, 83(1), 55-65.

Lindgren, M. (2004). Engine Exhaust Gas Emissions from Non-road Mobile Machinery. Department of Biometry and Engineering (Vol. PhD).

Lindgren, M. (2005). A transient fuel consumption model for non-road mobile machinery. Biosystems Engineering, 91, 139-147.

Lindgren, M., Hansson, P-A. (2004). Effects of transient conditions on exhaust emissions from two non-road diesel engines. Biosystems Engineering, 87, 57-66.

Lindvall, E.E., Törnquist, S., Enghag, O., Lundström, E., 2015. Biogasdrift i arbetsmaskiner. Slutrapport av regeringsuppdrag. Rapport 2015:23.

Lovarelli, D., Bacenetti, J., Fiala, M. (2017). Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. Journal of Cleaner Production, 140, 479–491.

Lovarelli, D., Bacenetti, J., Fiala, M. (2016). A new tool for life cycle inventories of agricultural machinery operations, Journal of Agricultural Engineering, XLVII, 40–53.

Marx, S.E., Luck, J.D., Hoy, R.M., Pitla, S.K., Blankenship, E.E., Darr, M.J. (2015). Validation of machine CAN bus J1939 fuel rate accuracy using Nebraska Tractor Test Laboratory fuel rate data. Computers and Electronics in Agriculture, 118, 179–185.

Molari, G., Mattetti, M., Perozzi, D., Sereni, E. (2013). Monitoring of the tractor working parameters from the CAN-bus. Journal of Agricultural Engineering, XLIV, 384-386.

Niero, M., Ingvordsen, C. H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M. F., Hauschild, M. Z., Jørgensen, R. B., 2015. Eco-efficient production of spring barley in a changed climate: A Life Cycle Assessment including primary data from future climate scenarios. Agricultural Systems, 136, 46–60.

Perozzi, D., Mattetti, M., Molari, G., Sereni, E. (2016). Methodology to analyse farm tractor idling time. Biosystems Engineering, 148, 81–89.

Pitla, S.K., Luck, J.D., Werner, J., Lin, N., Shearer, S.A. (2016). In-field fuel use and load states of agricultural field machinery. Computers and Electronics in Agriculture, 121, 290–300.

Renzulli, P.A., Bacenetti, J., Benedetto, G., Fusi, A., Ioppolo, G., Niero, M., Proto, M., Salomone, R., Sica, D., Supino, S. (2015). Life Cycle Assessment in the Cereal and Derived Products Sector. In: Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds). Life Cycle Assessment in the Agri-food Sector. Case Studies, Methodological Issues and Best Practices. Springer International Publishing, Switzerland. Pp. 185-249.

Sala, S., Anton, A., McLaren, S.J., Notarnicola, B., Saouter, E., Sonesson, U. (2017). In quest of reducing the environmental impacts of food production and consumption. Journal of Cleaner Production, 140, 387–398.

Shadidi, B., Yusaf, T., Alizadeh, H.H.A., Ghobadian, B. (2014). Experimental investigation of the tractor engine performance using diesohol fuel. Applied Energy, 114, 874-879.

Silva Capaz, R., Barreto Carvalho, V.S., Horta Nogueira, L.A. (2013). Impact of mechanization and previous burning reduction on GHG emissions of sugarcane harvesting operations in Brazil. Applied Energy, 102, 220-228.

Solaimuthu, C., Ganesan, V., Senthilkumar, D., Ramasamy, K.K (2015). Emission reductions studies of a biodiesel engine using EGR and SCR for agriculture operations in developing countries. Applied Energy, 138, 91-98.

Speckmann, H., Jahns, G., 1999. Development and application of an agricultural BUS for data transfer. Computers and Electronics in Agriculture, 23, 219-237.

Sørensen C.G., Halberg N., Ousdhoorn F.W., Petersen B.M., Dalgaard R. (2014). Energy inputs and GHG emissions of tillage systems. Biosystems Engineering, 120, 2-14.

Wolf M.A., Pant R., Chomkhamsri K., Sala S., Pennington D., 2012. International Reference Life Cycle Data System (ILCD) Handbook – Towards more sustainable production and consumption for a resource-efficient Europe. JRC Reference Report, EUR 24982 EN. European Commission – Joint Research Centre. Luxembourg. Publications Office of the European Union.

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Review

Water Footprint of crop productions: A review

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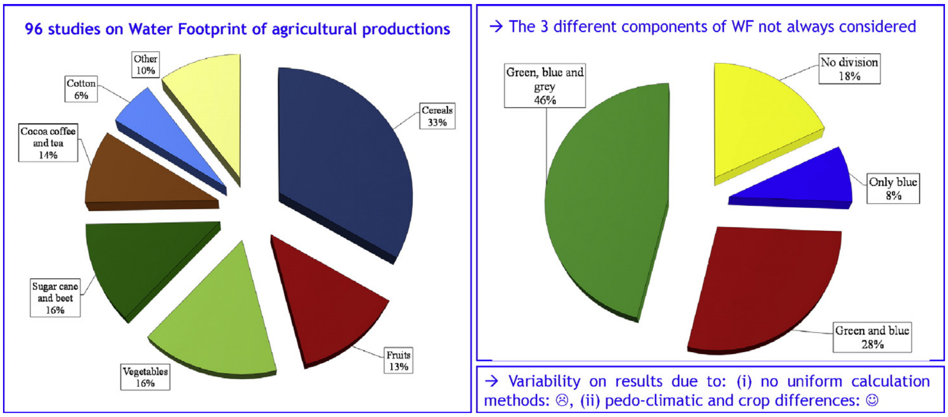
HIGHLIGHTS

- A literature review was completed on Water Footprint indicator.
- An advancement development of literature was followed.
- World and local studies with focus on agricultural productions were analysed.
- In 61% of studies a specific geographical area was considered.
- In 45% of studies grey water was assessed while in 18% only a total number was given.

GRAPHICAL ABSTRACT



→ Variability on results due to: (i) no uniform calculation methods: ☹, (ii) pedo-climatic and crop differences: ☺

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ABSTRACT

Water Footprint is an indicator recently developed with the goal of quantifying the virtual content of water in products and/or services. It can also be used to identify the worldwide virtual water trade. Water Footprint is composed of three parts (green, blue and grey waters) that make the assessment complete in accordance with the Water Footprint Network and with the recent ISO14046.

The importance of Water Footprint is linked to the need of taking consciousness about water content in products and services and of the achievable changes in productions, diets and market trades. In this study, a literature review has been completed on Water Footprint of agricultural productions. In particular, the focus was paid on crops for the production of food and bioenergy.

From the review, the development of the Water Footprint concept emerged: in early studies the main goal was to assess products' water trade on a global scale, while in the subsequent years, the goal was the rigorous quantification of the three components for specific crops and in specific geographical areas. In the most recent assessments, similarities about the methodology and the employed tools emerged.

For 96 scientific articles on Water Footprint indicator of agricultural productions, this literature review reports the main results and analyses weaknesses and strengths. Seventy-eight percent of studies aimed to quantify Water Footprint, while the remaining 22% analysed methodology, uncertainty, future trends and comparisons with other footprints. It emerged that most studies that quantified Water Footprint concerned cereals (33%), among which maize and wheat were the most investigated crops. In 46% of studies all the three components were assessed, while in 18% no indication about the subdivision was given; in the remaining 37%, only blue or green and blue components were quantified.

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Contents

1. Introduction

During the last years, high attention has started being paid on environmental analyses with multiple goals: quantifying environmental impacts of processes, identifying environmental hotspots and suggesting mitigation strategies to reduce the impact of anthropogenic productions on the environment.

Human impact on the environment has grown much more and faster than what was expected, and humanity consumes more resources (e.g., land, water) than what Earth is capable of regenerating (Galli et al., 2012; Hoekstra and Chapagain, 2008; IPCC, 2006). Immediate policies to limit the drawbacks and to restore a sustainable condition are needed, and stakeholders and decision makers are aware of this (Roelich et al., 2014; Wang et al., 2015). For example, more than 20% of Italian agricultural area is irrigated, but climate change is exposing the country to a deep change in precipitation trends (Natali et al., 2009). Thus the sector must adapt.

The most spread methodology to quantify the environmental impacts is the Life Cycle Assessment — LCA (ISO 14040, 2006) (Bacenetti et al., 2015a, 2015b; Bacenetti and Fusi, 2015; González-García et al., 2012; Ingrao et al., 2015a, 2015b; Rinaldi et al., 2014). Indicators such as Carbon Footprint, Ecological Footprint and Water Footprint have also developed to fulfil similar evaluations (Galli et al., 2012; Steen-Olsen et al., 2012) for specific environmental issues.

With regard to water, all over the world, the freshwater natural resource is getting precious, since scarcity and overexploitation are undeniable issues (Van Oel and Hoekstra, 2012; Zhang et al., 2013) that lead to social, environmental and economic problems (Ridoutt and Pfister, 2010). In more details, freshwater is a resource necessary not only for human and health concerns but also for productions and industrial processes; hence, its use must be distributed among different opportunities (e.g., Cazcarro et al., 2014; Lee, 2015). Because water is becoming scarcer and scarcer, mitigation strategies and a conscious use are key concerns.

In this context, a methodology was developed to analyse and quantify water use and to better understand the linkages between humanity's productive activities and the growing pressure on water directly and indirectly embedded in products and services (Hoekstra, 2010). This methodology is the “Water Footprint” (WF) and was introduced by Hoekstra and Hung (2002). Since then several studies have been carried out considering both the agricultural field production and the processing phases till the reach of consumers and waste disposal. Moreover, legislation to safeguard water has spread. WF was recently standardised by the ISO Standard 14,046 (ISO, 2014) and the EU defined the Water Framework Directive (WFD) (European Commission, 2010) to improve water quality, scarcity and productivity across Member States.

The aim of this paper is to carry out a literature review on the Water Footprint (WF) indicator, with focus on the WF of agricultural productions, and in particular of crops for food and energy purposes. The reason is that agricultural productions are the major responsible for water use and water stress (Hoekstra and Hung, 2002; Ridoutt and Pfister, 2010) and the availability of many studies inserted in different productive contexts needs clarity. In addition, even if WF has spread only in recent times, the concept upon which it grounds has gone through a constant progress; therefore, it is interesting to understand the aim and the development steps to comprehend its evolution.

The questions to which the present review aims to answer are:

(i) How did the concept of Water Footprint develop in the 10–15 years in which it started being used worldwide?
(ii) Is it a reliable indicator? Are there any limits to its application?
(iii) What are the limits of studies carried out till present?
(iv) How can its application and reliability be improved?

The outcomes of the present review can be helpful for policy makers and stakeholders in particular, in order to understand the usefulness of WF indicator and to develop policies and/or global decisions able to improve the freshwater use and to draft legislation on its sustainable consumption.

The paper is divided in five parts. In Section 2, WF approach and the definition of its components is given and in Section 3 the literature review of selected products is fulfilled. Finally, in Sections 4 and 5 WF limits and recommendations are analysed and conclusions are drawn.

2. Methods

2.1. Water Footprint definition

The concept of Virtual Water (VW) and the indicator of Water Footprint (WF) were developed over many years, and defined concepts and idea already clear in the 1990s. VW was first introduced by Allan (1997, 1998, 2001). It was defined as the water volume required to produce products or services during the production processes and not only the volume directly present in products (it is a “virtual” content). The concept got more precise and practical with Hoekstra and Hung (2002), Chapagain and Hoekstra (2003b), Hoekstra (2003), Oki et al. (2003), Zimmer and Renault (2003) and de Fraiture et al. (2004) who began quantifying the global virtual water flows. In particular, the VW concept was closely related to the Water Footprint one. This last was introduced to account for the appropriation of natural capital in terms of the water volumes required for human consumption (Hoekstra, 2010), in order to analyse all links between human consumption and water use (both directly and indirectly embedded in products and services) and between global trade and water resources management (Hoekstra and Chapagain, 2006).

The WF is commonly expressed as the water volume used to produce a unit of product (m^3/t) or the water volume per year of a delineated area (e.g., nation, province, catchment), individual or community (m^3/yr). When assessing the WF of a nation, market trades must be considered including products/services produced and consumed in the country, surpluses (water exported) and deficits (water imported) (Bulsink et al., 2010). Both internal and/or external national WF can be used to evaluate the market, the movement of water products and the sustainability of production and water use (Hoekstra, 2010; Steen-Olsen et al., 2012).

The WF indicator's relevance is due to consumers and producers that are often spatially disconnected from the production processes (Hoekstra and Chapagain, 2008; Hoekstra, 2010; Hoekstra et al., 2011). Quantifying the freshwater appropriation can raise awareness of actual water consumption.

To complete the framework of WF application, in addition to the original approach developed by the Water Footprint Network, a second recent approach has been developed (Vanham and Bidoglio, 2013). The two approaches are:

(i) the volumetric approach of the Water Footprint Network (WFN): it is the original approach developed according to Hoekstra and Hung (2002) and concerns the quantification of the virtual water content going through: (i) goals and scope, (ii) accounting, (iii) sustainability assessment, (iv) response formulation;
(ii) the LCA approach: it is in accordance with the recent ISO 14046 and similarly to LCA studies it must be carried out following: (i) goal and scope definition, (ii) inventory assessment, (iii) impact assessment, (iv) interpretation.

The most important difference between both methods is the product-focus of the LCA approach and the water management-focus of the WFN approach (Vanham, 2015). However, the methods have been compared by Manzardo et al. (2015) confirming the consistency in the results of both approaches, except for the degradative use process. In particular, a distinction between consumptive and degradative water use is given in Pfister et al. (2009, 2011). In the studies, authors describe the methodology to follow for the environmental impact of freshwater consumption and introduce the Water Stress Index (WSI). No matter which of the two approaches is chosen, the WF of a product, producer, consumer or nation gives coherent results. However, WF is not a single number stating the most or least water consuming production, but is made of three colour coded components: green, blue and grey water footprints (Hoekstra et al., 2011) and each of them represents an essential element of water use.

2.1.1. Green, blue and grey water

The WF of a product or service is the result of the quantification of three water volumes components:

(i) green (WF_{green}, m^3),
(ii) blue (WF_{blue}, m^3) and
(iii) grey (WF_{grey}, m^3).

The relative contribution of each of them to the total is important to be considered, because the meaning and role of each component are different (Hoekstra, 2013). The reference unit (Functional Unit — FU) must be defined according to the goal of the study. It can be expressed in different forms: tonne (t; for products) or year (yr; for individuals, communities or delineated geographical areas).

To produce a product of agricultural origin, the rainfall evaporated from soil, absorbed from roots and transpired by the crop and the water incorporated in harvested crop (Hoekstra et al., 2011) must be considered. This component is the "green water" and represents the major part of water commonly used during agricultural production phases.

It is usually calculated following Allen et al. (1998). It does not have any opportunity cost as it derives from natural processes and, consequently, only occurs when agricultural processes are part of the analysis. In particular, if evapotranspiration (ET) is higher than the effective rainfall occurred during crop growth (P), WF_{green} is equal to the effective rainfall (Eq. (1)); on the opposite, if evapotranspiration is lower than effective rainfall, WF_{green} is equal to evapotranspiration (Eq. (2)) (Bocchiola et al., 2013).

$$\text{if } ET \geq P_{eff}, WF_{green} = P_{eff} \tag{1}$$

$$\text{if } ET < P_{eff}, WF_{green} = ET \tag{2}$$

The second component is the "blue water" that is the surface or groundwater volume furnished on field through irrigation (i.e. evaporated water, product incorporated water and flow that does not return to the same catchment area) or that is pumped in the system during the processing phases (e.g., washings). In this component, water evaporated during and after irrigation is not included because it goes back to the natural system. Percolation water is a loss for the single field, but is not for the catchment area (Chapagain and Hoekstra, 2011). Blue water is expensive to use, since it has a high opportunity cost with other human activities; for example, Italian agriculture uses about 26 Gm^3/yr of water for irrigation purposes, which represents 49% of total water Italian needs (Antonelli and Greco, 2013). Reducing its use, both production costs (e.g., energy for pumping, machines and plants to buy and manage) and environmental impacts (due to energy, materials, plants, etc.) are reduced as well. With regard to the agricultural sector, the blue water used depends on the crop, on crop tolerance to water deficits, on irrigation efficiency and on green water. Whether green water is insufficient, blue water is used. In particular, if evapotranspiration is higher than rainfall, WF_{blue} is potentially equal to the difference between evapotranspiration and rainfall (Bocchiola et al., 2013). Otherwise, WF_{blue} is zero (Eq. (4)). In more details, Eq. (3) is valid when information about the irrigation method (e.g., surface, sprinkler and drip irrigation), schedule (e.g., irrigation turn, automatic irrigation) and volume, the soil water content, the crop growing features (e.g., root depth, crop cover) and the eventual water stress lack. On the opposite, when information about irrigation, soil and crop is available, more detailed WF_{blue} quantification is highly recommended (Hoekstra et al., 2011).

$$\text{if } ET \geq P_{eff}, WF_{blue} = ET - P_{eff} \tag{3}$$

$$\text{if } ET < P_{eff}, WF_{blue} = 0 \tag{4}$$

The WF_{blue} quantification can be carried out following water balance input–output assessment. This step is important because when a water stress occurs, yield is affected. The Relative Irrigation Supply (RIS) indicator described by García Morillo et al. (2015) can help to understand how irrigation matches the theoretical water requirement.

As regard to grey water, it is the volume necessary to assimilate the pollutants load caused by the production processes, based on existing ambient water quality standards (Hoekstra, 2010). It is not a real water volume used during production, but the volume needed to restore water quality after having been polluted along the production process.

Limits to water pollution and quality have been normed. It is important to reduce polluting products into water for multiple environmental reasons:

(i) no/less water gets polluted. This influences water and soil systems and ecosystems, enhancing environmental sustainability and public health (Mekonnen and Hoekstra, 2011; Watkins et al., 2006);
(ii) grey water reduction involves a reduction in water scarcity and in water competition, as less water must be restored in its quality (Liu et al., 2012; Mekonnen and Hoekstra, 2011; Orlowsky et al., 2014).

According to Hoekstra (2010), WF_{grey} (m^3/yr) is calculated as:

$$WF_{grey} = \frac{L_{add}}{C_{max} - C_{nat}} \tag{5}$$

where:

L_{add} (Gg/yr) pollutant load,
C_{max} (mg/dm^3) ambient water quality standard (i.e. *maximum* acceptable concentration for the pollutant),
C_{nat} (mg/dm^3) natural concentration of the pollutant in the receiving water body.

L_{add} is:

$$L_{add} = L - C_{nat} \cdot Q_{act} \tag{6}$$

where:

L (Gg/yr) total pollutant (or nutrient) load
Q_{act} (m^3/yr) actual basin discharge.

Commonly, the natural concentration is not zero, because all rivers naturally transport some nutrients. However, it is often considered zero for simplicity, for keeping a conservative approach (Aldaya and Hoekstra, 2010; Mekonnen and Hoekstra, 2011) and when human-made pollutants are considered (Franke et al., 2013). In several studies, the grey component is not assessed, due to difficulties in evaluating the pollutants and in integrating the component in real water volumes (Bocchiola et al., 2013; Jefferies et al., 2012; Mekonnen et al., 2015b; Vanham, 2015). WF_{grey} is predominantly due to nutrient load of fertilisers used in agriculture, sewage and industrial wastewaters (Liu et al., 2012; Mekonnen and Hoekstra, 2015). For agricultural productions, nitrogen is commonly responsible for WF_{grey} because nitrogen fertilisers are the most abundantly spread on field. For each substance, the maximum permissible limits vary according to local conditions; either the United States Environmental Protection Agency (US-EPA) limits or the EU recommended limits are used. As regard to nitrogen, US-EPA recommends using 45 mg/dm^3 of NO_3, whereas the EU recommends using 50 mg/dm^3 of NO_3. Still, WF reliability and robustness is affected by the lack of proper data that can wrongly affect results (Bulsink et al., 2010; Galli et al., 2012; Hoekstra and Chapagain, 2008). Franke et al. (2013) drew comprehensive guidelines for WF_{grey} accounting in which three methods of increasing detail (Tier 1, Tier 2 and Tier 3) were defined. In addition, since the WF_{grey} effect depends on the discharge to assimilate the pollutant in the catchments, the water pollution level (WPL) can be quantified as well. It measures the fraction of pollution assimilation capacity of a river basin that is consumed (Hoekstra et al., 2011). When water is polluted, it is considered not usable (Brown and Matlock, 2011).

The three components described (green, blue and grey) must be quantified separately; nevertheless, in order to have a single indicator, they can be summed in a single number (Eq. (7)).

$$WF = WF_{green} + WF_{blue} + WF_{grey} \tag{7}$$

The choice for a single indicator often causes discussions among researchers, because their economic, environmental and social impacts are different (e.g., opportunity costs, land use, water depletion). According to the economic sector of reference, the relative contribution of the three components differs; in particular, a distinction can be made among industrial, domestic and agricultural water use. In the industrial and domestic sectors, the highest shares are attributed to blue water, while in the agricultural one, green water has a consistent role. Moreover, the agricultural sector has the highest global contribution to water use (>90%, Mekonnen and Hoekstra, 2010).

For green and blue waters, Brown and Matlock (2011) identified social and environmental impacts. On the social point of view, use of land modifies the availability of green and blue water for other purposes and for future generations. On the environmental point of view, impacts concern (i) a change in use of land from natural to agricultural land and a loss of natural ecosystems and biodiversity and (ii) a competition in the use of irrigation water with other industrial uses.

2.2. Methodology adopted

This paper reviews 96 scientific studies on the Water Footprint of agricultural productions published till November 2015. Referring to them, Fig. 1 shows the number of studies published per year from 2000 to 2015: the number of studies increased in the most recent years, proving the growing interest on the indicator. Moreover, the higher number of studies is linked to a higher specificity as well. It also emerges that most of studies till 2008 were carried out on agricultural productions with a world extent. Because WF is a recently developed indicator, those studies represent the methodological basis for calculating WF and give average values valid for worldwide productions. Moreover, the focus is on global virtual water trades (imports and exports) occurring for crops. From 2009, an increasing number of studies emerged, focused either on one or few crops cultivated on a specific location or on all crops that characterised an area. The greater detail for crops is due to the availability of a defined methodology and to the consciousness that local climate and local productive choices affect WF.

Thus, in the next Section the literature analysis of crops WF follows. In more details, this review is organised grouping studies in which similar crops were assessed (with both a world extent and a local extent for single or small groups of agricultural productions) and basing the structure of this study on the advancement obtained in the WF field. The review focuses on the agricultural production phases of crops for food, feed and fibre products and of bioenergy biomass. The agricultural sector was analysed because it is the major responsible for freshwater use (Mekonnen, 2011) and products of vegetable origin are at the basis of the food and feed chain.

3. State of the art on the Water Footprint of agricultural productions

Table 1 reports the studies analysed in the present literature review.

3.1. WF on a world extent

Hoekstra and Hung (2002) were the first to make a global estimate of the consumptive water use for several crops in several countries, but did not split the composition in green, blue and grey water. From here on, numerous studies have been carried out on a worldwide scale (Chapagain et al., 2006; Chapagain and Hoekstra, 2011; Mekonnen and Hoekstra, 2011). According to Chapagain and Hoekstra (2004), the water volume globally used for crops production in the period 1997–2001 was 6390 Gm^3/yr. Generally, the virtual water content of crops was lower than animal products, as these last include also water for feed production (Enne et al., 2006; Gerbens-Leenes et al., 2013; Hoekstra and Hung, 2002; Hoekstra, 2003; Mekonnen and Hoekstra, 2012; Oki et al., 2003; Palhares and Pezzopane, 2015; Pahlow

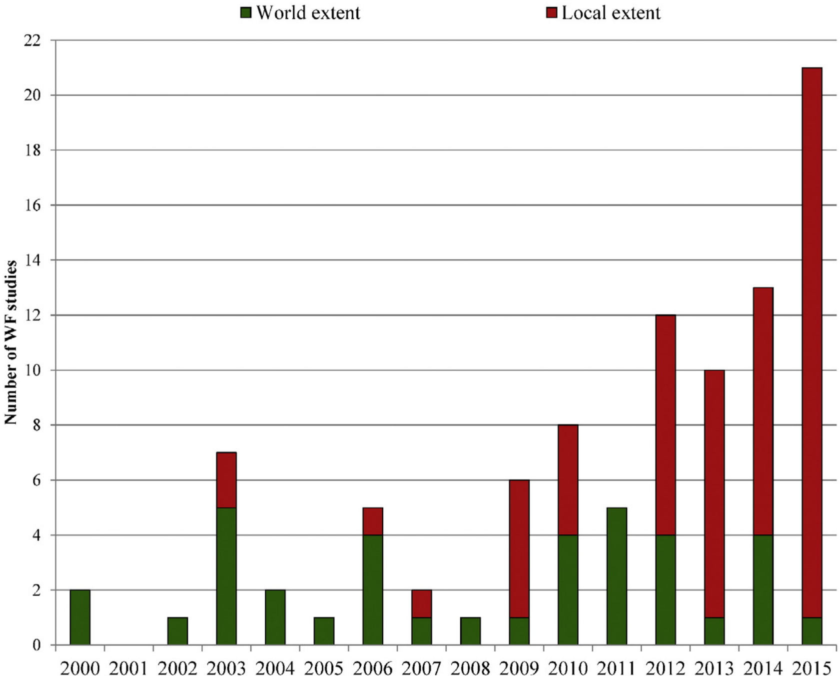

Fig. 1. Number of studies on WF indicator per year (2000–2015) that have been considered in the present review and that have either a world or a local extent.

et al., 2015; Vanham et al., 2013). The average WF quantified by Chapagain and Hoekstra (2004) was 900, 1300 and 3000 m^3/t for maize, wheat and rice, respectively. Instead, for animal products such as chicken meat, pork and beef, it was 3900, 4900 and 15,500 m^3/t, respectively. The main concern about these quantifications was that they provided a global average not valid in every context, since geographic and temporal locations, climate, technology and yield are local issues. Recent studies that quantified the WF of animal productions supported these evaluations; for example, Palhares and Pezzopane (2015) quantified WF of dairy farms comparing conventional and organic farming (1422 m^3/kg ECM milk and 1510 m^3/kg ECM milk, respectively); Pahlow et al. (2015) quantified the WF of farmed fishes and crustaceans equal to 1974 m^3/t (83% green, 9% blue and 8% grey) and de Miguel et al. (2015) quantified the Spanish WF for pork production and processing, equal to 19.5 billion m^3/yr (82% green, 8% blue and 10% grey).

Chapagain (2006) evaluated virtual water trades trying to understand fluxes of water embedded in products and the impact of import from water-scarce or water-abundant countries, either for resource lack of the country or for saving domestic water in the country. The possibility of increasing the dependency from water-abundant countries to reduce local freshwater depletion was supported by a 222 Gm^3/yr saving, obtained through better water trade distribution. However, introducing water savings by increasing dependency from external trades had drawbacks, also highlighted by Fader et al. (2011), such as the risk of (i) reducing food self-sufficiency, (ii) reducing employment in the agricultural sector, (iii) increasing the environmental impact of exporting–importing products and (iv) not importing as much as needed, requiring to produce anyway the same products. Such risk was underlined by Chapagain and Hoekstra (2008) as well, who reported that 16% of total worldwide WF was attributable to products coming from the external market.

Mekonnen and Hoekstra (2010) and Hoekstra et al. (2011) studied the WF of a multitude of products and processes assessing agriculture at a high spatial resolution. For example, among crops, maize had the lowest WF (1222 m^3/t) while wheat (1827 m^3/t) had the highest and rice stood near the average (average for crops: 1644 m^3/t). The result on rice was similar to Chapagain and Hoekstra (2010), where it was 1675 m^3/t. Sugar crops and vegetables showed low WF (200 and 300 m^3/t, respectively). Fruits reached 1000 m^3/t and oil crops 2400 m^3/t. Pulses, spices and nuts required higher volumes, varying between 4000 and 9000 m^3/t, respectively.

Nevertheless, the uncertainty of studies and the lack of spatial and geographical specificity caused inaccuracy. Therefore, the need of carrying out local studies grew faster.

The first attempt to modify the global water trade analyses was carried out in 2008 by Hoekstra and Chapagain (2008) when they argued that green, blue and grey waters should not be summed to a total WF because their opportunity costs and impacts were different. In particular, they also started focusing on concerns such as double counting, water returns to catchments and runoff.

Even more importance of taking into account separately the three components arose with Mekonnen (2011) who showed that in the period 1996–2005 the global WF was 9087 Gm^3/yr (30% higher than what assessed by Chapagain and Hoekstra in 2004). The allocation of the three components was 74% green, 11% blue and 15% grey. Distinguishing among sectors, agriculture resulted contributing for 91% of this total; the remaining 5% was attributed to industrial goods and 4% to domestic use. The average global consumer had a WF equal to 1385 m^3/yr, but US citizens almost consumed the double (2842 m^3/yr per capita). On the opposite, most African countries had the lowest WFs (500–600 m^3/yr per capita). With regard to crops, for example, the global wheat production required 1088 Gm^3/yr (70% green, 19% blue and 11% grey).

Regarding grey water, an interesting study was carried out by Liu et al. (2012) who calculated the critical substance responsible for pollution, the WF_{grey} and the WPL in worldwide rivers using the Global NEWS (Global Nutrient Export from WaterSheds) model. Results showed that for the 1000 river basins analysed, about two thirds had an undesirable WPL (WPL > 1). More in details, in year 2000, rivers with WPL > 1 were 11% for N pollution and 54% for P pollution.

To identify and define the WF role when combined with the other Footprints, Steen-Olsen et al. (2012) studied carbon, land and water footprints of EU27. Referred to 2004, the WF_{blue} (the only component assessed, in accordance with Ridoutt and Pfister, 2010) resulted 87 Gm^3/yr in the EU, which corresponded to 179 m^3/yr per capita (10% above the global average). Italy was slightly above the EU27 average; in particular, because of climate, Italy was one of the most

Table 1
Main results of the literature review.

Products	Area	Period	FU	Phases assessed	Component*	Author
Agricultural commodities	Morocco and Netherlands		yr		No dist.	Hoekstra and Chapagain (2006)
Agricultural, industrial and domestic sectors	Latin America and Caribbean	1996–2005	yr		Green, blue, grey	Mekonnen et al. (2015a)
Agriculture (multiple crops, e.g., maize, cassava, wheat, oilseeds), industry and domestic	World — 16 regions (USA, Canada, Japan and South Korea, Western Europe, Eastern Europe, Australia and New Zealand, Russia, Middle East, Central America, South America, South Asia, South-East Asia, China, North Africa, Sub-Saharan Africa and rest of the world)	2000 and 2050	yr		Green, blue, grey	Ercin and Hoekstra (2014)
Agriculture, industry and domestic	World	1997–2001	yr		No dist.	Chapagain and Hoekstra (2008)
Agriculture, industry, domestic and waste management	Netherlands		yr		Blue	Hoekstra et al. (2012)
Agriculture, industry and domestic	China	2007	yr		No dist.	Dong et al. (2014)
Agriculture, industry and domestic	China	2007	yr		No dist.	Dong et al. (2013)
Almonds, barley, carrots, dates, figs, grapes, olives, oranges, potatoes, tomatoes and wheat	Tunisia	1996–2005	t, yr and m^3	Cultivation	Green, blue, grey	Chouchane et al. (2015)
Almonds, barley, grapes, maize, olives, oranges, sugar beets, sugar cane, mandarins, tomatoes, wheat	Morocco		yr	Cultivation	Green, blue, grey	Schyns and Hoekstra (2014)
Bagasse and rice straw for second generation bioethanol	Taiwan		dm^3 EtOH	Raw material production, processing (refining)	Green, blue, grey	Chiu et al. (2015)
Bioenergy and biofuel biomass	World (USA, Brazil, China, Germany, Italy, India, France, UK, Pakistan South Africa, etc.)	2030		Processing	Blue	Gerbens-Leenes et al. (2012)
Bioethanol	USA	2005–2008	t		Blue	Chiu et al. (2009)
Bioethanol (sugarcane, sugar beets, maize)	World		t	Cultivation and processing	Green, blue, grey	Gerbens-Leenes and Hoekstra (2012)
Cassava	Thailand, Vietnam, Colombia		t	Processing and wastewater treatment	Not expl.	Tran et al. (2015)
Coal, lignite, natural gas, oil, uranium, wood, wind, solar, geothermal, and hydropower	World (Europe, China, USA and Canada, Latin America and Caribbean, Africa, India)	2008–2012	TJ	Power plant production and operation	Green, blue	Mekonnen et al. (2015b)
Cotton	World — 15 countries (USA, Brazil, China, India, Mali, Pakistan, Uzbekistan, EU, etc.)	1997–2001	t and yr	Cultivation and consumption	Green, blue, grey	Chapagain et al. (2006)
Cotton, soybean, animal products, cocoa, coffee, maize, barley, fodder crops, wheat, rapeseed, sunflower, potatoes, etc.	France		yr	Cultivation, processing and use	Green, blue, grey	Ercin et al. (2013)
Coarse cereals, wheat, rice, maize, sunflower	China	1960–2010			Not expl.	Liu et al. (2015)
Crops (wheat, rice, maize, barley, sorghum, millet, cassava, potatoes, sweet potatoes, sugar cane, sugar beet, soybeans, groundnuts, sunflower seeds, soybean oil, groundnut oil, sunflower seed and oil, etc.)	World	1997–2001	yr		Green, blue	Yang et al. (2006)
Crops and animal products	World	1971–2000 and 2070–2099	yr and capita	Cultivation and production	Green, blue	Gerten et al. (2011)
Crops and animal products	World	1996–2005	t	Feed, production, drinking water, service water, feed-mixing water	Green, blue, grey	Mekonnen and Hoekstra (2012)
Crops for food, feed	Cyprus	1995–2009	yr		Green, blue	Zoumides et al. (2014)
Durum wheat and pasta	Italy, USA, Greece, Turkey	2011	kg	Cultivation, processing, packaging and transport	Green, blue, grey	Ruini et al. (2013)
Maize	Italy	2001–2010	mm	Cultivation	Green, blue	Bocchiola et al. (2013)
Maize	Italy	2001–2010	kg	Cultivation	Green, blue	Nana et al. (2014)
Maize, *Miscanthus*, poplar, potatoes, sunflower	Netherlands, USA, Brazil, Zimbabwe		GJ	Cultivation and	Green, blue	Gerbens-Leenes

(continued on next page)

Table 1 (*continued*)

Products	Area	Period	FU	Phases assessed	Component*	Author
and wheat for energy purposes				processing		et al. (2009)
Maize, palm oil, raw sugar cane, barley, cotton, rice, coffee, cocoa, wheat, soybean, groundnuts, potatoes, millet, cassava and sorghum etc.	World	1997–2001	t and yr	Cultivation	No dist.	Chapagain and Hoekstra (2004)
Maize, wheat, vegetables, melons, oil plants, cotton	China	1978–2012	yr		Green, blue, grey	Xu et al. (2015)
Maize, soybean, rice and wheat	Yellow river basin	1996–2005	t	Cultivation	Green, blue	Zhuo et al. (2014)
Maize, soybean, wheat, and rice plus industrial, domestic, energetic and livestock sectors	World (USA, China, Latin America, Europe, Asia)		ha		Blue	Elliott et al. (2014)
Maize, wheat, tropical crops	Egypt, Ethiopia, India, China, Australia, Argentina, France, USA		t		Green, blue	Hoff et al. (2010)
Maize, wheat	China	1960–2008	kg	Cultivation	Green, blue	Sun et al. (2013)
Multiple crops	World	1995–1999	yr		No dist.	Hoekstra and Hung (2002)
Maize, wheat, soybean, sweet potato, groundnut, watermelon and vegetables	China	2002–2008	t		Green, blue and grey	Huang et al. (2012)
Multiple (e.g., rice, wheat, maize)	World	1995	m^3, yr		Green, blue	de Fraiture et al. (2004)
Multiple (e.g., fruits, meat, bread, cheese, coffee, tea, cotton, vegetables, groundnuts, paper, sugar, wine, biodiesel)	World	2010	kg, cup 125 ml, glass 250 ml, 1 sheet 80 g/m^2		No dist.	Hoekstra (2010)
Multiple (wheat, maize, potatoes, beets, cane, vegetables, citrus, fruits, groundnuts, etc.)	World (China, Indonesia, USA, EU)		ha	Cultivation	Green, blue	Bruinsma (2003)
Multiple (wheat, fodder crops, barley and maize)	EU28	2012	yr		Green, blue	Vanham and Bidoglio (2013)
Multiple (wheat, rice, barley, maize etc.)	World				Green, blue, grey	Mekonnen and Hoekstra (2014)
Multiple crops	World	1996–2005	yr	Cultivation	Green, blue, grey	Mekonnen (2011)
Multiple crops	World	1997–2001	yr		Green, blue, grey	Chapagain (2006)
Multiple crops	World				Green, blue, grey	Mekonnen and Hoekstra (2010)
Multiple (126 primary crops)	World	1996–2005	t		Green, blue, grey	Mekonnen and Hoekstra (2011)
Olives	Italy	2009–2014	t	Cultivation	Green, blue, grey	Pellegrini et al. (2015)
Oranges and strawberries	Brazil, China, USA, Italy, Spain, Morocco, UK and Poland		kg and ha	Cultivation	No dist.	Mordini et al. (2009)
Pasta sauce and peanuts	Australia		575 g sauce and 250 g peanuts	Cultivation, processing, packaging and consumption	Green, blue, grey	Ridoutt and Pfister (2010)
Potatoes	Great Britain	1981–2010	t		Blue	Hess et al. (2015)
Potatoes	Argentina		t	Cultivation	Green, blue, grey	Rodriguez et al. (2015)
Primary, processed, transformed products and by-products	World	1961–1999	kg and yr	Cultivation and processing	Not expl.	Zimmer and Renault (2003)
Rice	South Korea	2004–2009	t and yr	Cultivation	Green, blue, grey	Yoo et al. (2013)
Rice	China, India, Indonesia, Bangladesh, Vietnam, Thailand, Myanmar, Philippines, Brazil, Japan, USA, Pakistan, Korea	2000–2004	yr	Cultivation	Green, blue, grey	Chapagain and Hoekstra (2010)
Rice	Argentina	2009–2013	t	Cultivation	Green, blue	Marano and Filippi (2015)
Rice	Global (China, India, Indonesia, Bangladesh, Vietnam, Thailand, Myanmar, Japan, the Philippines, Brazil, USA, Pakistan, Korea rep., etc.)	2000–2004	yr and t	Cultivation and consumption	Green, blue, grey	Chapagain and Hoekstra (2011)
Rice soybeans and wheat	World — 92 countries	2030				Konar et al. (2013)
Rice, maize, cassava, soybeans, groundnuts, coconuts, oil palm, bananas, coffee, cocoa	Indonesia	2000–2004	t	Cultivation	Green, blue, grey	Bulsink et al. (2010)
Rice, maize, sweet potatoes, sugarcane and sweet sorghum	Taiwan	2007–2011	t and dm^3 EtOH	Cultivation and processing (refining)	Green, blue, grey	Su et al. (2015)

Table 1 (*continued*)

Products	Area	Period	FU	Phases assessed	Component*	Author
Rice, wheat, soybean, maize, barley, beef, pork, chicken, egg	Japan	1996–2000	t		No dist.	Oki et al. (2003)
Rice, wheat, maize, miscellaneous cereals, beans and tubers	China	2010	kg	Cultivation	Green, blue	Wang et al. (2014)
Strawberry	Spain	2010–2012	t		Blue	García Morillo et al. (2015)
Sugar beet	Europe		Not expl.	Cultivation and processing (refining)	Green, blue, grey	Thaler et al. (2012)
Sugar cane	Brazil		t		Green, blue, grey	Scarpare et al. (2016)
Tea and coffee	Netherlands	1995–1999	125 ml coffee cup; 250 ml tea cup	Cultivation and consumption	Not expl.	Chapagain and Hoekstra (2007)
Tea and margarine	Kenya, Indonesia and India, Germany	2005–2007	50 g tea and 500 g margarine	Cultivation, transport packaging and distribution	Green, blue	Jefferies et al. (2012)
Temperate cereals, rice, maize, tropical cereals, pulses, temperate roots, tropical roots, sunflower, soybeans, groundnuts, rapeseed	World	1998–2002	t		Green, blue	Fader et al. (2011)
Tomato	Mediterranean		kg	Cultivation	Not expl.	Antón et al. (2005)
Tomato	Spain			Cultivation	Green, blue, grey	Chapagain and Orr (2009)
Tomato, bread wheat, durum wheat, pasta and pizza	Italy		t	Cultivation and processing	Green, blue, grey	Aldaya and Hoekstra (2010)
Tomato sauce	USA		680 g	Cultivation, processing, packaging	Green, blue, grey	Manzardo et al. (2015)
Tomato dried		2011	kg	Cultivation	Green, blue	Ramírez et al. (2015)
Wheat	World	1996–2005	yr and t	Cultivation and consumption	Green, blue, grey	Mekonnen and Hoekstra (2010)
Wheat, rice, maize, soybeans and potatoes	China	1998–2010		Cultivation	Green, blue	Cao et al. (2015)
Wheat, rice, legumes, maize, other cereals, potatoes, bread, snacks, biscuits, vegetables, fruits, dried fruits, pork, lamb, beef, poultry meat, aquatic products, other meats, eggs, milk, butter, yogurt, cheese, sugar, sweets, vegetable oils and others	China	2004, 2006, 2009	Per capita	Food waste	No dist.	Song et al. (2015)
Wine	Italy	2011	dm^3	Cultivation and processing	Green, blue, grey	Lamastra et al. (2014)

* Note: "not expl." stands for "not explicit" and "no dist." stands for "no distinction".

water consuming European countries (Antonelli and Greco, 2013; Hoekstra et al., 2011). Also Galli et al. (2012) studied the three main Footprints and stated that Water Footprint was the indicator with the lowest overlapping with the two other indicators and that European countries were the most important virtual water importers. Similar conclusions were drawn by Vanham and Bidoglio (2013) who completed a review on WF of EU28. Results showed that Italy and Spain accounted for two thirds of the European irrigation water withdrawal (WF_{blue}). Italy, Spain, Portugal, Greece and France together accounted for 96% of EU WF_{blue} and France, Spain, Italy, Poland, Germany and Romany were the most important green water consumers. The virtual water content of products was low in northern and western EU, while southern and eastern EU had the highest. The reason was the amount and type of agricultural products consumed, as well as the local climate. Specifically for the Spanish agricultural sector, Duarte et al. (2014) quantified WF and confirmed that Spain was one of the largest water depleting countries.

According to Mekonnen and Hoekstra (2014), most of former literature values on a worldwide scale were not enough representative for the global variability and could not be used to draw clear conclusions. From a statistical analysis emerged that some former literature data were too much approximated and that WF had a lower impact than what was attributed. Moreover, they quantified the global water saving whether green, blue and grey water were reduced (even reaching 79% reduction for grey water). A closed result was obtained by Mekonnen et al. (2015a) in a study focused on Latin America and the Caribbean countries.

3.2. WF for specific and local agricultural productions for food, feed and fibre

As previously mentioned, studies in specific contexts and specific agricultural crop productions developed in a second phase.

The first study analysed concerning crops cultivation was by Chapagain et al. (2006) who calculated WF of cotton consumption assessing the most 15 cotton-productive countries. Differently from former studies (Chapagain and Hoekstra, 2003b, 2004; Hoekstra and Hung, 2002), the three components were calculated. The result was about 9800 Gm^3/yr. The lowest WF was in USA and Brazil where green water was enough to have low blue water. Except for China and USA, most countries were water-intensive cotton producers. WF_{grey} had a remarkable role and was calculated in each processing stage using the permissible limits recommended by US-EPA (2005). The virtual water flow between countries was 204 Gm^3/yr (40% green, 43%

blue and 17% grey); 84% of EU cotton-related WF mainly lied in India (with a high WF_{blue} share, equal to 5.75 Gm^3/yr).

Crops analysed by Bulsink et al. (2010) were responsible for 86% of total Indonesian water use. For WF_{grey}, US-EPA (2005) nitrogen limit was used. Results showed cassava having the lowest WF (514 m^3/t). Coffee was the most impacting (22,907 m^3/t) because of both high green water (96%) and high grey water (4%). The second largest resulted cocoa, with a WF equal to 9414 m^3/t (94% green, 0% blue and 6% grey). Nevertheless, considering the total surface cultivated with these crops, the highest WF was still rice one: 3473 m^3/t (73% green, 21% blue and 6% grey). The largest virtual water flows were mainly attributed to oil palm, coffee and coconut oil. In the Netherlands, coffee and tea production and consumption were studied by Chapagain and Hoekstra (2003a, 2007): WF was 140 dm^3/cup of coffee and 34 dm^3/cup of tea. Concerning tea, Jefferies et al. (2012) studied WF referred to 50 g tea produced in Kenya, Indonesia and India, transported to UK and packed and distributed in Belgium. In the same study, they also assessed the WF of 500 g margarine produced and sold in Germany. Green and blue water were assessed through CROPWAT. With regard to tea, the WF calculated by Chapagain and Hoekstra (2003a, 2007) and Mekonnen and Hoekstra (2010) was not comparable to the one by Jefferies et al. (2012) because both the Functional Unit and the system boundary were different.

Concerning tomato production, Chapagain and Orr (2009) studied the WF analysing Spanish water resources use both in open field and in greenhouses. Greenhouses were the most spread over Spain with 60,000 ha (14% of the horticulture area). CROPWAT was used to calculate green water in open field, while crop evaporation dynamics were different in greenhouses, where, according to indoor climatic data, evapotranspiration was assumed 70–80% of that in open fields. Nitrogen was the grey water indicator, with the limit set by the EU (50 mg $N\text{-}NO_3/dm^3$). On average, WF_{grey} was 7.2 m^3/t. Results were quite different from other data available in literature: Antón et al. (2005) had WF 4 times lower, with a study on both LCA and WF; Chapagain and Hoekstra (2004) had much higher results (+60%) but they overestimated green water in covered systems because the different evapotranspiration dynamic was not assessed; Aldaya and Hoekstra (2010) analysed Italian industrial tomato production that had a WF equal to 114 m^3/t (30% green, 50% blue and 17% grey). Comparing these results with Chapagain and Orr (2009), the blue component was in accordance with them, while green and grey were much higher in the Italian productive system, due to different weather conditions and fertilisers inputs. Referred to 680 g of sauce, Manzardo et al. (2015) quantified in 125 dm^3 the tomato cultivation WF in USA (1% green, 72% blue and 34% grey) plus 7.6 dm^3 for the processing to sauce (22% blue and 78% grey). Referred to 1 kg dried tomato, Ramírez et al. (2015) quantified WF including cultivation (85 dm^3/kg with no green water use; 37 dm^3/kg with collector — lower blue and grey water use) and drying (15 dm^3/kg). Page et al. (2012) assessed WF of tomato production in Australia comparing open field to greenhouse production with low, medium and high-tech, inserting the study in the LCA context. Med-tech greenhouses were responsible for the highest WF (52 dm^3/kg), while open field and high-tech greenhouses had the lowest value (5.0 and 5.4 dm^3/kg, respectively).

With an LCA perspective, also WF of oranges and strawberries was assessed by Mordini et al. (2009). China had the largest content of Virtual Water (VW) per kg oranges, followed by Spain, Italy and Brazil, whereas oranges produced in the USA had the lowest. When comparing the VW in m^3/ha, the differences were due to yield (7.5 t/ha and 38.7 t/ha in China and USA, respectively), as a similar water volume was used in most countries (i.e. USA and China, approximately 5000 m^3/ha). With regard to strawberries, Poland had the highest water consumption in terms of m^3/kg, while Morocco and Spain had the highest WF in m^3/ha. Although UK and Poland had a similar water consumption per hectare (about 2500 m^3/ha), the consumption per kg was lower in UK, due the higher yield. About strawberries, a study was completed by García Morillo et al. (2015), who quantified the average WF (70 m^3/t) for Spanish cultivation. Differences in water use depended on soil typology, protected or unprotected systems, irrigation system, country of origin and climate, yield and farming system (i.e. organic or conventional).

Concerning rice, an assessment was carried out by Chapagain and Hoekstra (2010) who studied freshwater consumption for the 13 most important rice producing countries (more than 90% of global production, average yield 4.49 t/ha). Irrigation and rainfall were calculated and grey water was quantified using nitrogen as representative element, differently from former studies where WF_{grey} was excluded and from Chapagain and Hoekstra (2004) where an average constant percolation loss (300 mm) was added to green and blue waters. On average, rice WF was 1325 m^3/t (48% green, 44% blue ad 8% grey). This corresponded to 1391 Gm^3/yr, which was closed to the global rice WF by Chapagain and Hoekstra (2004). Yoo et al. (2013) calculated Korean rice WF equal to 844.5 m^3/t that was a slightly different result from Chapagain and Hoekstra (2010), where it was quantified equal to 829 m^3/t. Given the similar assumptions, the results were probably influenced by climate in the different temporal contexts. Also Marano and Filippi (2015) calculated rice WF in Argentina with a similar result: 845 m^3/t (43.5% green and 56.3% blue) in Northern regions and 987 m^3/t (36.5% green and 63.5% blue) in Southern. Given the climate differences, the biggest variability was shown by blue water use.

Recently, several studies have been carried out on China and its provinces. The major attention was paid on the low water availability in most Chinese provinces that undermines agricultural cultivations for food production (Cao et al., 2015; Dong et al., 2014; Song et al., 2015). With 70% of total water use, agriculture was the most water consuming sector in the country (Dong et al., 2013). However, there were considerable disparities among provinces, showing ranges between most (Southern regions; 27.5 billion m^3/yr in Guangdong) and least (Northern regions; 2.72 billion m^3/yr in Qinghai) water availabilities (Dong et al., 2014). In more details, Song et al. (2015) calculated the average water consumption of Chinese population analysing 27 vegetable and animal products; water consumption was equal to 673 m^3/yr per capita and primarily affected by pork meat and rice production. To this amount 18 m^3/yr per capita had to be added as waste water from food waste. In 2014, Wang et al. (2014) completed a study in which the highest WF was attributed to rice production (1.36 m^3/kg), while the lowest to maize (0.91 m^3/kg). Sun et al. (2013) quantified the WF of maize and wheat in the Hetao province, where rainfall lack affected crops cultivation. Thus, green water lack was replaced by blue water, which was about 91% of total WF for the two assessed crops. Moreover, authors showed that the improvement of irrigation efficiency positively affected the blue water use (from 9.25 m^3/kg in 1960 to 0.79 m^3/kg in years 2000). Blue water had a considerable role on products WF (Xu et al., 2015). Similar results were obtained by Liu et al. (2015), who assessed WF for cereals in the same province. In their study, the increased WF from 1960 to 2010 was quantified through an increase tendency during the '80s and a decrease during years 2000. Cao et al. (2015) quantified the water use and water productivity for Chinese provinces concerning wheat, rice, maize, soybeans and potatoes. Accordingly, 67% of farms were irrigated and used 68% of national water. Results were expressed for all crops and water use was 43% blue water and 57% green water. With regard to rice production, the average Chinese WF was similar to the average by Chapagain and Hoekstra (2004), while it was higher than in Yoo et al. (2013). Concerning maize, instead, WF in Chinese provinces was lower than the global average. For example, in Huang et al. (2012) maize WF was 868 m^3/t (48.5% green, 0.5% blue, 51.0% grey). However, maize was particularly affected by local climate and the water lack deeply affected yield. This concept was highlighted by Bocchiola et al. (2013) who focused on Italian maize grain production, evaluating the impact of climate change scenarios on yield and WF in Italian Po Valley. They assessed temperatures increase (+2 to +6 °C) with an effect on

CO_2 concentrations (+10 to +30%), on rainfall (−20% to +20%), duration of the growing season and yield (average 11.43 t/ha). Climate change scenarios were made according to simulation Global Circulation Models (GCMs). Focusing on the worst cases and similarly to the conclusions by Hoekstra et al. (2011), Ercin and Hoekstra (2014) and Elliott et al. (2014) significant WF_{blue} increases affected the sustainability of productions because yield reductions were prominent when the climate was unfavourable. Concerning the same area, Nana et al. (2014) identified a model for WF_{green} and WF_{blue} quantification and applied it to maize grain in Po Valley (Italy), with results equal to 479 kg/kg (41% green and 59% blue).

In their study about processed products, Aldaya and Hoekstra (2010) assessed the WF of Italian pasta and pizza. To achieve the results, they quantified wheat WF making a distinction between durum wheat (*Triticum durum*) and bread wheat (*Triticum aestivum*): durum wheat had an average WF much higher than bread wheat (+50%): WF_{green} and WF_{blue} of durum wheat were 1.5 and 4.2 times higher than bread one, respectively. WF_{grey} was 1.8 times bigger in durum rather than in bread wheat (considering 10 mg/dm^3 of NO_3-N from US-EPA as limit). Comparing these results with Mekonnen and Hoekstra (2011), it emerged that in Italy, both durum and bread wheat WF had a different allocation on the three components: blue and grey waters were higher in the Italian rather than in the global average (Italian durum wheat: +53% and +45% for blue and grey components, respectively; Italian bread wheat: +17% and +10% for blue and grey components, respectively). The green one, on the opposite, was lower (66% and 39% of the global average for durum and bread wheat, respectively). Italian wheat WF (both durum – 1574 m^3/t – and bread – 786 m^3/t – especially) lied in a lower position than the global crops average (1644 m^3/t; Mekonnen and Hoekstra, 2011). Also Huang et al. (2012), quantified wheat WF in Beijing (China) with an even lower result: 712 m^3/t (31% green, 53% blue, 16% grey).

Ruini et al. (2013) calculated WF of pasta production. Results showed that the most impacting phases were durum wheat cultivation (on average, 1.644 dm^3/kg of pasta) and cooking (10 dm^3/kg of pasta). Processing and packaging had much low WF. In particular, pasta WF ranged between 1.336 dm^3/kg and 2.847 dm^3/kg when produced in Italy and Turkey, respectively. WF_{green} ranged between 72% of total WF in USA (offset by the high blue water) and 91% in Turkey. WF_{grey} had a considerable role on the production.

The WF of different economic Moroccan activities was assessed in Schyns and Hoekstra (2014). Being agriculture responsible for the major share (77%) of national WF, the focus was paid on agricultural productions. Bread wheat had the largest WF per year (11 Gm^3/yr). The largest WFs were associated to the lowest economic productions values (0.08 US\$/m^3 for wheat). On the opposite, with the exception of almonds, the lowest surfaces (e.g., 0.1 million ha for tomatoes) were attributed to crops with the highest economic value (e.g., 1.81 \$/m^3 for tomatoes) and low WFs. For example, WF of tomatoes, oranges and mandarins was 0.10 Gm^3/yr, 0.44 Gm^3/yr and 0.20 Gm^3/yr, respectively. Consequently, authors stated it would be worthwhile on the water-economic point of view, to reconsider crops produced in Morocco. Results by Schyns and Hoekstra (2014) could not be compared with other studies because the functional units (FUs) were different (e.g., t, yr) and, when the year was indicated as FU, it referred to a different geographical (different amount of hectares and/or yield) and temporal area. For example, Chouchane et al. (2015) studied the Tunisian WF, but results were not comparable. Crop economic values were used to quantify the water productivity (WP; t/m^3; ratio between agricultural output and water used) and the economic water productivity (EWP; \$/m^3; crop value per unit of water used). Almonds showed the highest WF (20,820 m^3/t, which was twice more than the global average) distributed as 85.3% green, 9.3% blue and 5.3% grey water. On the opposite, tomatoes had the smallest WF (120 m^3/t) with a share among green, blue and grey water equal to 50.0%, 41.6% and 8.4%, respectively. Almonds and olives showed the highest WF_{green} (85% and 96% of their total WF, respectively) while dates and figs showed the highest WF_{blue} (74% and 37% of total WF, respectively). Almonds also had the highest WF_{grey} (5%), followed by barley and figs (4% and 3%, respectively). Moreover, almost all crops had a higher WF in Southern regions if compared to WF of those cultivated in Northern and Central ones. Similarly, Zoumides et al. (2014) calculated WF of several crops cultivated in Cyprus considering their economic value. Results highlighted that during the years in which crops had a higher economic value, the blue water use was higher. In particular, a high average blue (13%) and green (87%) water use emerged from Cypriote productions. Benefits from high-value productions carried out with blue water emerged on the economic point of view, at the expense of the environmental one. The semi-arid climate and the local water scarcity affected WF with a consistent dependency from external trades.

About potatoes, Hess et al. (2015) quantified WF for Great Britain: on average it was 75 m^3/t (85% green, 15% blue). The result was lower than in Chouchane et al. (2015), where it was 250 m^3/t (44% green, 48% blue and 8% grey), confirming the discrepancies in water use between North and South countries. On the opposite, Argentinian potatoes WF was quantified by Rodriguez et al. (2015) and resulted equal to 324 m^3/t (56% green and blue, 44% grey water). In Huang et al. (2012), Chinese sweet potato production had a WF equal to 823 m^3/t (59% green, 29% blue, 12% grey), which was considerably higher than other accountings, and probably due to local climate and agricultural management. In their study, also other crops were quantified and WF resulted the highest for soybean (1816 m^3/t, 66% green, 19% blue, 15% grey) and for groundnuts (1330 m^3/t, 66% green, 19% blue, 15% grey). For watermelon, on the opposite, WF was the lowest (136 m^3/t, 29% green, 32% blue, 39% grey). For vegetables grown in open or covered system, the difference was +73 m^3/t blue water in the covered one.

As regard to wine, vineyard production and winery processing were analysed in Lamastra et al. (2014). They took into account the three water footprint components and showed that grey water had a significant role on wine production in Italian system (about 17% of total wine WF). On the opposite, irrigation was rarely carried out; therefore, blue water contributed only for 7% to wine WF. The Italian WF for wine ranged between 745 and 1084 dm^3/dm^3 wine, while the Italian average quantified by Mekonnen and Hoekstra (2011) was 697 dm^3/dm^3 wine, which underestimated of 7%–46% the effective value. Another traditional Italian orchard system was olive production. Pellegrini et al. (2015) quantified olive WF in three systems (i.e. traditional, intensive and high-density) and results showed that the traditional system had the highest WF (3434 m^3/t; 64% green, 24% blue and 12% grey), while the high-density had the lowest (2782 m^3/t; 25% green, 72% blue and 3% grey). These values were +14% and −8% the global average (3015 m^3/t).

3.3. WF for specific and local agricultural productions with energy purposes

In addition to the food, feed and fibre use, crops can also be cultivated for bioenergy purposes. In view of this scope, several studies have been carried out to quantify the WF of a bioenergy use of crops.

Among these, Gerbens-Leenes et al. (2009) quantified the WF of energy from biomass and compared it with the one from fossil energy. Being agriculture the most water depleting sector, biomass production undeniably increased WF. Therefore, even if renewable energy production is supported thanks to CO_2-neutrality and supply security, biomass energy production heavily impacts on water use. Among the Netherlands, USA, Brazil and Zimbabwe, the Netherlands had the least WF (2069 m^3/t) equal to 24 m^3/GJ. Zimbabwe, on the contrary, had the largest (142 m^3/GJ). In more details, the lowest WF was obtained from Dutch maize and wheat (9 m^3/GJ for both), from American sugar beets (30 m^3/GJ) and from Brazilian and Zimbabwe's sugarcane (25 m^3/GJ and 31 m^3/GJ, respectively).

Concerning bioethanol production, already Chiu et al. (2009), Gerbens-Leenes et al. (2009) and King and Webber (2008) made an assessment on water consumption. However, Gerbens-Leenes and Hoekstra (2012) assessed WF of first generation bioethanol in a wider extent, considering sugar cane and beets (61% of bioethanol production) and maize (39% of bioethanol production). WF for sugar cane processing ranged between 1.0 and 21.0 m^3/t, while the one for sugar beet between 0.0 and 4.5 m^3/t. The global average WF was 209 m^3/t, 133 m^3/t and 1222 m^3/t, for sugar cane, sugar beet and maize, respectively.

Su et al. (2015) also studied crops for bioethanol production in Taiwan. For most crops, WF_{green} exceeded 50% of total WF. The highest WF was due to rice (average 2404.5 m^3/t), while the lowest WF was attributed to sweet potatoes (on average, 107.0 m^3/t). Also Chiu et al. (2015) carried out a study on bioethanol production in Taiwan. However, they studied second generation production including raw materials production and processing. Results showed a lower WF than Su et al. (2015). In particular, raw rice for second generation had a WF equal to 920.4 m^3/t (15% green, 81% blue and 4% grey) and sugar cane to 34.5 m^3/t (87% green, 10% blue and 3% grey). To this amount the operational WF had to be added: 1% and 16% of the total WF of blue and grey water, respectively, for rice straw, and 72% and 93% of the total WF of blue and grey water, respectively, for the bagasse.

Biomass for bioenergy production by Mekonnen et al. (2015b) aimed to compare different fossil and renewable energy sources. WF was calculated for multiple sources, among which wood. On a global average, wood had the highest WF per unit of electricity (156,000 m^3/TJ).

Scarpare et al. (2016) quantified sugarcane WF in Brazil, WF was 98 m^3/t of sugar cane (58% green, 20% blue and 12% grey) in irrigated fields and 119 m^3/t (85% green and 15% grey) in rain-fed ones.

Tran et al. (2015) assessed WF of starch production from cassava for bioenergy uses, in addition to carbon footprint and energy use. However, the cultivation phase was excluded. WF ranged between 10 and 60 m^3/t according to the plant typology.

3.4. WF in future trend scenarios

Scenarios for future trends in water uses were carried out in multiple studies (Ercin and Hoekstra, 2014; Gerten et al., 2011; Orlowsky et al., 2014) in order to consider the freshwater scarcity and pollution that will affect water availability and quality. According to all of them, human dependency on freshwater will increase and problems for future food security and environmental sustainability will increase as well (Alcamo et al., 2003; Bruinsma, 2003; Rosegrant et al., 2007). In particular, it is thought that in 2025 about 67% of the global population will experience water scarcity and 15% absolute water scarcity (less than 500 m^3/yr per capita) (Alcamo et al., 2000; Galli et al., 2012; Vörösmarty et al., 2000).

Other studies on future scenarios were carried out by Gerten et al. (2011), where water stress was expressed following different drivers (climate, population, diets) showing significant reductions in water availability. Hoekstra et al. (2012) analysed the seasonal dependence attributed to water scarcity, while Gerbens-Leenes et al. (2012) analysed blue water scarcity in bioenergy and biofuel production presenting a scarcity increase equal to 5.5% by 2030. Elliott et al. (2014) studied the climate change impact on irrigation, asserting that the conversion from rain-fed to irrigated agriculture will not be enough to offset the climate change effect. Considering climate, also Konar et al. (2013) modelled future trades, focusing on rice, soybeans, wheat and an aggregate of crops, while Nelson et al. (2014) used a model to investigate economic responses in agriculture. Scenarios till 2050 were assessed by Ercin and Hoekstra (2014). They took into account drivers such as population, economic growth, production/trade patterns, diets and bioenergy uses, climate, social behaviour, policy, and technological development. Results showed WF increase at different steepness, according to the drivers considered. However, a WF reduction was possible, provided that other drivers changed at population growth. A similar result was obtained by Orlowsky et al. (2014). Even given the limitations of their study, reduced water availability will reduce WF of some nations and will entail reduced export capacity and effects on consumption of other countries.

4. Discussion

Fig. 2 shows the subdivision among analysed agricultural productions (e.g., cereals, fruits, vegetables), among which cereals are the most investigated crops (33%). Fig. 3 shows the subdivision among cereals (e.g., maize, rice, wheat) according to their presence in studies in which these crops were analysed. (See Fig. 3.)

Of the 96 studies about agricultural productions, 75 had as primary goal the WF quantification either on a global or on a local extent; 14 aimed at studying the future implications of water use and water scarcity and of WF uncertainty; 2 studies analysed the indicator and the available data on a statistical point of view, while 3 aimed at realising a specific literature review; 2 more studies analysed WF in the context of the “Footprints Family”, identifying and comparing Carbon, Ecological and Water Footprints. Focusing on the 75 articles where crops WF was quantified, results showed that 62% of them studied a local or defined geographical area, while the remaining 38% were carried out on a world extent. In all of them, WF_{green} and WF_{blue} were quantified, while only in 46% authors took WF_{grey} into account. In more details, in 18% of papers there was no statement whether WF_{grey} was assessed or not. Fig. 4 shows the subdivision of studies according to the assessed components. When WF_{grey} was quantified, only nitrogen was considered for water quality restoration. Two methods were mostly used for the definition of water quality limits, US-EPA (45 mg/dm^3 of NO_3-N) and EU permissible limit (50 mg/dm^3 of NO_3-N). In 25% of studies in which WF_{grey} was quantified, US-EPA N-permissive limit was used; in 8%, the EU permissive limit was used, while in 67%, no indication was given. In particular, the method selected for the permissible limits was not explicitly defined in studies with a global scale, in which, however, according to Laane et al. (2005) and to Liu et al. (2012), limits should be locally defined.

WF_{green} and WF_{blue} were always quantified. In 49% of studies, the model used to assess WF_{green} was CROPWAT by FAO. In the remaining ones, either the model was not declared or in 8% of articles, the model was different and it was stated the name of the model used (i.e. Cropsyst, MRIO, Lund-Potsdam-Lena, Waterstat, PCR-Globwb).

In the first studies carried out till around 2008, no distinction was made among green, blue and grey water volumes. In the following

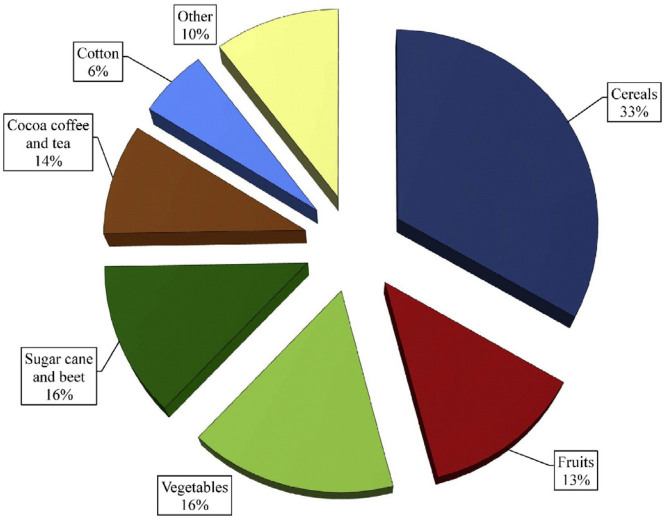

Fig. 2. Subdivision of crops analysed in the reviewed studies.

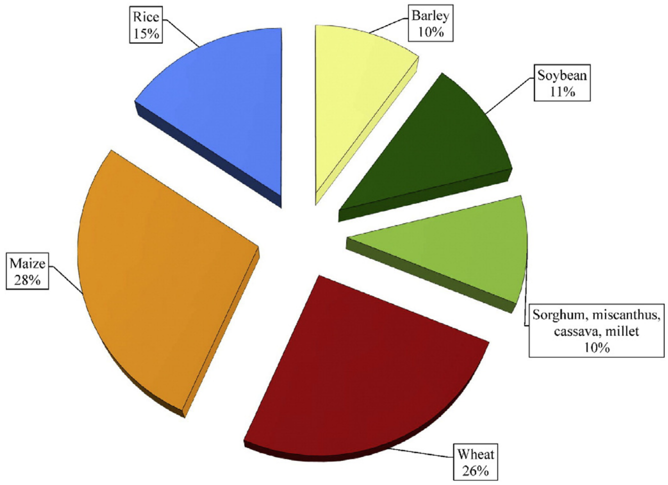

Fig. 3. Subdivision of cereal crops analysed in the reviewed studies.

years, the three components were separated from each other, making achievable a complete analysis of water volumes with different economic, social and environmental effects, considering benefits and drawbacks (e.g., comparison between organic and conventional farming systems). In addition, the spatial distribution of countries and their geographical and geomorphological structure should be considered in efficiency evaluations. The results of studies present in this literature review are often incomparable either because not referred to a comparable functional unit (e.g., yr) or because the geographical and/or temporal location differ and different assumptions were made.

4.1. Variability in the results

In several studies (Elliott et al., 2014; Ercin and Hoekstra, 2014; Hoff et al., 2010; Mekonnen and Hoekstra, 2010, 2011; Schyns and Hoekstra, 2014; Yang et al., 2006; Zhuo et al., 2014) the high uncertainty (±20%) of WF assessments emerged. Uncertainty is a consequence of the several variables that characterise the agricultural sector and that must be taken into account. Considering agricultural crops production, the major contributor to WF is green water, which is subjected to rainfall seasonality and which affects blue water. This explains the uncertainty that characterises big geographical areas. Moreover, uncertainty can increase if virtual water trades are considered, because water uses in each country must be analysed as function of other countries.

A higher specificity linked to crops, cultivation techniques, regions and periods should be investigated to limit uncertainty. In more details, crops WF deeply depends on the agricultural management techniques (Xu et al., 2015); thus, policies to promote more sustainable pesticides application, soil tests, slow-release fertilisers and high efficiency irrigation solutions can help to reduce water stress (Herath et al., 2014). Moreover, highly-educated farmers are more likely to adopt environmental-friendly policies (e.g., crop diversification, improved seeds and cultivation techniques), which can determine beneficial effects on water use.

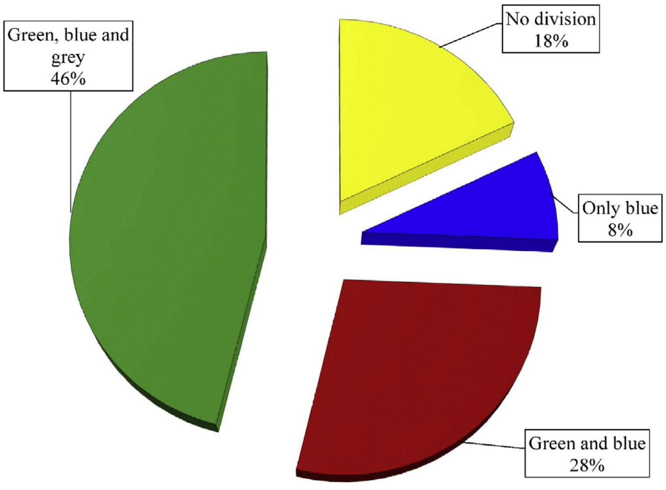

Fig. 4. Subdivision of reviewed studies according to the components considered.

4.2. Identified limits in WF assessment

Many studies were carried out on WF assessment, but some methodology differences arose. Actually, too often studies aggregate the three components green, blue and grey, without contextualising the water use. Climatic variables may force producers to produce with too much blue water, but maybe what they grow is the only or one of the only crops adequate to those climatic conditions. It is important not to forget the agronomic features such as soil texture, pH and nutrient and mineral soils content, which can deeply affect the producer choice, as well as the local water scarcity, the economic return and the global market policies. Moreover, the commercial global trade cannot only be varied according to water use and water scarcity. On the other hand, however, WF can support decision making and the introduction of technologies that allow for consistent water volume reductions (Pellegrini et al., 2015).

Studies often quantify blue water without information about the irrigation method and schedule, which affects the result. In particular, the irrigation method (i.e. surface, sprinkler or drip irrigation) involves different irrigation efficiencies. Moreover, frequently farmers do not have constantly water for irrigation purposes, but undergo irrigation turns. This influences the water stress and the water deficit to which the crop is exposed, and finally the yield. To take into account this information and its consequences, for example, García Morillo et al. (2015) introduced the concept of the total volume of water applied for crop growing (CWA_{blue}; m^3/ha) that allowed for considering the real irrigation volumes. Similarly, Scarpare et al. (2016) for sugarcane production in Brazil, Yoo et al. (2013) for paddy rice in Korea and Cao et al. (2015) for maize grain in China used the total irrigation volume to assess the WF_{blue}.

With regard to grey water, literature still shows several studies lacking of it. However, it is important to take into account water pollution and water quality in order to not underestimate the problem. In addition, a drawback is that WF is not completely assessed according to methodology and Standards. In particular, quantifying only crops green and blue water use is closed to their water balance, not to water footprint. The spread lack in WF_{grey} assessment is motivated by authors that criticise the loss of information by introducing a component that evaluates water quality “virtually” embedded and not water quantity (Thaler et al., 2012; Vanham, 2015). For example, Hastings and Pegram (2012) state that WF_{grey} does not consider eco-toxicity, biodegradability or water treatment neither. WF_{grey} strongly depends on the assumptions made about elements included in the quantification (nitrogen, phosphorous, metals and pesticides). These critiques are founded on correct basis because what emerges from the use of Hoekstra et al. (2011) for WF_{grey} assessment is a simplification and limitation. In particular, when the WF is applied to agricultural production, WF_{grey} should be calculated following a more detailed methodology and using field estimations and referring to the most specific Tier III. The pollutant concentration released in the water bodies should be estimated considering site-specific parameters such as daily precipitation, field slope, soil characteristics (e.g., texture, carbon content) and run-off. In this regard, a first interesting approach was developed by Lamastra et al. (2014). In the study, authors developed a detailed calculation method for the WF_{grey} quantification, which is a valid option for studies carried out on crops production. In particular, they considered the predicted environmental concentration, the pesticide's toxicity and the dilution factor for vineyard production considering site-specific parameters. A similar methodology applicable in multiple productive contexts should be developed for detailed WF_{grey} assessments, which would allow for locally defined studies and for predictions of potential policies and/or behaviours.

Another issue is that some countries may do their best in water use efficiency, but still have high water consumption (e.g., lack of technology). As reported in Hoekstra (2013), Mekonnen and Hoekstra (2014) and Pfister and Bayer (2014), it would be useful to make WF evaluations considering the best available technology and practise (e.g., optimised nutrients management, crop rotation, crop residues management, precision irrigation, reduction of non-beneficial evapotranspiration and effective rainfall enhancement), but this condition does not cover the real farming contexts. Local conditions, countries capabilities and technologies are the major features affecting water volumes and water use efficiency for agricultural productions. Moreover, convenience evaluations (opportunity-costs in particular) for water use should be considered as well when local consumptions and global appropriations are studied (Vanham and Bidoglio, 2013).

These considerations were done in studies aiming to suggest possible improvements to reduce WF; in more details, possible mitigating solutions are (Hoekstra et al., 2011):

(i) reduction of domestic use and food waste;
(ii) increase of green and blue water productivity for agricultural products;
(iii) reduction of fertilisers and pesticides use and their more effective use;
(iv) citizens' consumption adaptations, especially of animal products.

Similarly to the other Footprints, WF should not be seen as an ultimate sustainability indicator, but as one to accomplish the sustainability debate (Chapagain et al., 2006). In response to its usefulness, it does not give any new knowledge, but gives a new perspective on water scarcity issues, water dependency, sustainable use and global trade implications for water management. Nevertheless, Wichelns (2015) is sceptical upon this indicator and concluded that WF is much different from Carbon and Ecological Footprint and that its role must be carefully analysed. In particular, the author reported that WF scope has changed from its initial development framework: WF was firstly assessed for virtual water trades worldwide, while recently literature has changed, attributing to a high WF a high impact on the water resource and on its depletion and vice versa. In his study he reported four perspectives concerning VW and WF, which are:

(i) WF should not modify the international trade, regarding trade from water-abundant to water-scarce countries and technology and opportunity costs of producers and markets that often are not taken into account;
(ii) engaging in VW trade does not save water in countries, because the water savings estimate does not have any policy relevance since producers' decisions do not shift according to VW import and the available water in a water-scarce country will be totally used in any case;
(iii) water scarcity/quality issues in one country cannot be solved by consumers of other countries, because water scarcity and quality are local problems and cannot be analysed attributing to other consumers the responsibility of scarcity and quality;
(iv) Carbon and Ecological Footprints differ from Water Footprint; Carbon Footprint, for example, indicates that whether CO_2-eq is reduced, the pressure on the atmosphere will be reduced as well, while the same statement is not valid for WF. The Ecological Footprint answers to a global scale, indicating pressure on land, while water is a local issue and WF gives no information about environmental impact.

Regardless, Ercin et al. (2013), Fader et al. (2011) and Vanham and Bidoglio (2013) reported a positive remark of WF application, which is:

(i) with regard to WF_{blue}, even if in water-abundant countries there is no scarcity in water, a more efficient water use can be useful for: (i) increasing productions using the same amount of water, (ii) reducing WF_{blue} in water-scarce areas, because less import from water-scarce countries is necessary if a higher production is made in water-abundant countries, (iii) water can be allocated for producing other goods;
(ii) with regard to WF_{green}, the need of reducing it, even if it does not have any costs, derives from the fact that it is also a scarce resource to preserve.

5. Conclusions

A literature review of Water Footprint indicator was carried out with the goal of making clarity in the available literature. The focus was paid on crops for the food chain and for bioenergy purposes. The review was dealt with grouping similar crops and following an advancement order of literature. Both WF studies with a world and local extent were taken into account.

The result of the study is that WF was submitted to progresses in the last decade and its methodology and goals moved from global analyses about water trades to local detailed analyses of water volumes directly and indirectly embedded in products. Moreover, literature showed some critiques about WF use, especially when decision-making seemed to be considered dependent on the amount of water entailed in products. WF indicator cannot be used on its own as measure of productions sustainability, because the local, pedo-climatic and technological context must be considered. In particular, also the three components, green, blue and grey water cannot be considered on the same economic, social and environmental level. Focusing on market trades, blue water is the most important for decision making issues, because it represents a direct cost for society; instead, grey water has considerable importance on the environmental point of view.

In conclusion, some improvements can still be achieved in WF calculation and are required to make this indicator more valid for crop production contexts.

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References

Alcamo, J., Henrichs, T., Rösch, T., 2000. World-water in 2025 — global modeling and scenario analysis for the world commission on water for the 21st century. Report A0002, Center for Environmental Systems Research, University of Kassel, Kurt Wolters Strasse 3, 34109, Kassel, Germany.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S., 2003. Global estimates of water withdrawals and availability under current and future “business-as-usual” conditions. Hydrol. Sci. J. 48 (3), 339–348. http://dx.doi.org/10.1623/hysj.48.3.339.45278.

Aldaya, M.M., Hoekstra, A.Y., 2010. The water needed for Italians to eat pasta and pizza. Agric. Syst. 103 (6), 351–360. http://dx.doi.org/10.1016/j.agsy.2010.03.004.

Allan, J.A., 1997. ‘Virtual water’: a long term solution for water short Middle Eastern economies? Occasional Paper 3. School of Oriental and African Studies (SOAS), University of London

Allan, J.A., 1998. Virtual water: a strategic resource, global solutions to regional deficits. Groundwater 36 (4), 545–546.

Allan, J.A., 2001. The Middle East water question: hydropolitics and the global economy. I.B. Tauris, London.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration: guidelines for computing crop water requirements. FAO Drainage and Irrigation Paper 56. Food and Agriculture Organization, Rome.

Antón, M.A., Montero, J.I., Muñoz, P., 2005. LCA and tomato production in Mediterranean greenhouses. Int. J. Agric. Resour. Gov. Ecol. 4 (2), 102–112.

Antonelli, M., Greco, F., 2013. L'acqua che mangiamo. Cos'è l'acqua virtuale e come la consumiamo. Edizioni Ambiente.

Bacenetti, J., Fusi, A., 2015. The environmental burdens of maize silage production: influence of different ensiling techniques. Anim. Feed Sci. Technol. 204, 88–98.

Bacenetti, J., Duca, D., Fusi, A., Negri, M., Fiala, M., 2015a. Mitigation strategies in the agro-food sector: the anaerobic digestion of tomato puree by-products. An Italian case study. Sci. Total Environ. 526, 88–97.

Bacenetti, J., Fusi, A., Negri, M., Fiala, M., 2015b. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Bocchiola, D., Nana, E., Soncini, A., 2013. Impact of climate change scenarios on crop yield and water footprint of maize in the Po valley of Italy. Agric. Water Manag. 116, 50–61. http://dx.doi.org/10.1016/j.agwat.2012.10.009.

Brown, A., Matlock, M.D., 2011. A review of water scarcity indices and methodologies. The Sustainability Consortium, 19. White Paper n.106.

Bruinsma, J., 2003. World Agriculture: Towards 2015/2030: An FAO Perspective. Earthscan, UK, London, p. 97.

Bulsink, F., Hoekstra, A.Y., Booij, M.J., 2010. The water footprint of Indonesian provinces related to the consumption of crop products. Hydrol. Earth Syst. Sci. Discuss. 6 (4), 5115–5137. http://dx.doi.org/10.5194/hessd-6-5115-2009.

Cao, X., Wang, Y., Wu, P., Zhao, X., Wang, J., 2015. An evaluation of the water utilization and grain production of irrigated and rain-fed croplands in China. Sci. Total Environ. 529, 10–20. http://dx.doi.org/10.1016/j.scitotenv.2015.05.050.

Cazcarro, I., Hoekstra, A.Y., Sánchez Chóliz, J., 2014. The water footprint of tourism in Spain. Tour. Manag. 40, 90–101. http://dx.doi.org/10.1016/j.tourman.2013.05.010.

Chapagain, A.K., 2006. Globalisation of Water: Opportunities and Threats of Virtual Water Trade. A.A. Balkema Publishers, Taylor & Francis Group plc, Leiden.

Chapagain, A.K., Hoekstra, A.Y., 2003a. The water needed to have the Dutch drink coffee. Water Research Series 14(14). UNESCO-IHE, Delft, The Netherlands, p. 8.

Chapagain, A.K., Hoekstra, A.Y., 2003b. Virtual water trade. Virtual water flows between nations in relation to trade in livestock and livestock products. International Expert Meeting on Virtual Water Trade 13(11), pp. 1–202.

Chapagain, A.K., Hoekstra, A.Y., 2004. Water footprints of nations. Value of Water Research Report Series No. 16. UNESCO-IHE, Delft, The Netherlands.

Chapagain, A.K., Hoekstra, A.Y., 2007. The water footprint of coffee and tea consumption in the Netherlands. Ecol. Econ. 64 (1), 109–118.

Chapagain, A.K., Hoekstra, A.Y., 2008. The global component of freshwater demand and supply: an assessment of virtual water flows between nations as a result of trade in agricultural and industrial products. Water Int. 33 (1), 19–32.

Chapagain, A.K., Hoekstra, A.Y., 2010. The green, blue and grey Water Footprint of rice from both a production and consumption perspective. Water Research Report Series No. 40. UNESCO-IHE, Delft, The Netherlands.

Chapagain, A.K., Hoekstra, A.Y., 2011. The blue, green and grey water footprint of rice from production and consumption perspectives. Ecol. Econ. 70 (4), 749–758. http://dx.doi.org/10.1016/j.ecolecon.2010.11.012.

Chapagain, A.K., Orr, S., 2009. An improved water footprint methodology linking global consumption to local water resources: a case of Spanish tomatoes. J. Environ. Manag. 90 (2), 1219–1228. http://dx.doi.org/10.1016/j.jenvman.2008.06.006.

Chapagain, A.K., Hoekstra, A.Y., Savenije, H.H.G., Gautam, R., 2006. The water footprint of cotton consumption: an assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. Ecol. Econ. 60, 186–203. http://dx.doi.org/10.1016/j.eco lecon.2005.11.027.

Chiu, Y.W., Walseth, B., Suh, S., 2009. Water embodied in bioethanol in the United States. Environ. Sci. Technol. 43 (8), 2688–2692. http://dx.doi.org/10.1021/es8031067.

Chiu, C.C., Shiang, W.-J., Lin, C.J., Wang, C.-H., Chang, D.-M., 2015. Water footprint analysis of second-generation bioethanol in Taiwan. J. Clean. Prod. 101 (1000), 271–277. http://dx.doi.org/10.1016/j.jclepro.2015.03.068.

Chouchane, H., Hoekstra, A.Y., Krol, M.S., Mekonnen, M.M., 2015. The water footprint of Tunisia from an economic perspective. Ecol. Indic. 52, 311–319. http://dx.doi.org/10.1016/j.ecolind.2014.12.015.

de Fraiture, C., Cai, X., Amarasinghe, U., Rosegrant, M., Molden, D., 2004. Does international cereal trade save water? The impact of virtual water trade on global water use. Comprehensive Assessment Research Report vol. 4.

de Miguel, Á., Hoekstra, A.Y., García-Calvo, E., 2015. Sustainability of the water footprint of the Spanish pork industry. Ecol. Indic. 57, 465–474. http://dx.doi.org/10.1016/j.ecolind.2015.05.023.

Dong, H., Geng, Y., Sarkis, J., Fujita, T., Okadera, T., Xue, B., 2013. Regional water footprint evaluation in China: a case of Liaoning. Sci. Total Environ. 442, 215–224. http://dx.doi.org/10.1016/j.scitotenv.2012.10.049.

Dong, H., Geng, Y., Fujita, T., Fujii, M., Hao, D., Yu, X., 2014. Uncovering regional disparity of China's water footprint and inter-provincial virtual water flows. Sci. Total Environ. 500-501, 120–130. http://dx.doi.org/10.1016/j.scitotenv.2014.08.094.

Duarte, R., Pinilla, V., Serrano, A., 2014. The water footprint of the Spanish agricultural sector: 1860–2010. Ecol. Econ. 108 (2014), 200–207. http://dx.doi.org/10.1016/j.ecolecon.2014.10.020.

Elliott, J., Deryng, D., Müller, C., Frieler, K., Konzmann, M., Gerten, D., ... Wisser, D., 2014. Constraints and potentials of future irrigation water availability on agricultural production under climate change. Proc. Natl. Acad. Sci. U. S. A. 111 (9), 3239–3244. http://dx.doi.org/10.1073/pnas.1222474110.

Enne, G., Greppi, G., Serrantoni, M., 2006. Il ruolo dell'acqua nell'allevamento animale. Ital. J. Agron. 3, 519–527.

Ercin, A.E., Hoekstra, A.Y., 2014. Water footprint scenarios for 2050: a global analysis. Environ. Int. 64, 71–82. http://dx.doi.org/10.1016/j.envint.2013.11.019.

Ercin, A.E., Mekonnen, M.M., Hoekstra, A.Y., 2013. Sustainability of national consumption from a water resources perspective: the case study for France. Ecol. Econ. 88, 133–147. http://dx.doi.org/10.1016/j.ecolecon.2013.01.015.

European Commission, 2010. EU water framework directive. http://ec.europa.eu/environment/pubs/pdf/factsheets/water-framework-directive.pdf (Available online).

Fader, M., Gerten, D., Thammer, M., Heinke, J., Lotze-Campen, H., Lucht, W., Cramer, W., 2011. Internal and external green-blue agricultural water footprints of nations, and related water and land savings through trade. Hydrol. Earth Syst. Sci. 15 (5), 1641–1660. http://dx.doi.org/10.5194/hess-15-1641-2011.

FAO, 2015. CROPWAT. http://www.fao.org/nr/water/infores_databases_cropwat.html (Accessed 27 September 2015).

Franke, N.A., Boyacioglu, H., Hoekstra, A.Y., 2013. Grey water footprint accounting. Water Research Report Series No. 65. UNESCO-IHE, Delft, The Netherlands.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, B., Giljum, S., 2012. Integrating ecological, carbon and water footprint into a "footprint family" of indicators: definition and role in tracking human pressure on the planet. Ecol. Indic. 16, 100–112. http://dx.doi.org/10.1016/j.ecolind.2011.06.017.

García Morillo, J., Rodríguez Díaz, J.A., Camacho, E., Montesinos, P., 2015. Linking water footprint accounting with irrigation management in high value crops. J. Clean. Prod. 87, 594–602. http://dx.doi.org/10.1016/j.jclepro.2014.09.043.

Gerbens-Leenes, P.W., Hoekstra, A.Y., 2012. The water footprint of sweeteners and bio-ethanol. Environ. Int. 40 (1), 202–211. http://dx.doi.org/10.1016/j.envint.2011.06.006.

Gerbens-Leenes, P.W., Hoekstra, A.Y., van der Meer, T.H., 2009. The water footprint of energy from biomass: a quantitative assessment and consequences of an increasing share of bio-energy in energy supply. Ecol. Econ. 68 (4), 1052–1060. http://dx.doi.org/10.1016/j.ecolecon.2008.07.013.

Gerbens-Leenes, P.W., van Lienden, A.R., Hoekstra, A.Y., van der Meer, T.H., 2012. Biofuel scenarios in a water perspective: the global blue and green water footprint of road transport in 2030. Glob. Environ. Chang. 22 (3), 764–775. http://dx.doi.org/10.1016/j.gloenvcha.2012.04.001.

Gerbens-Leenes, P.W., Mekonnen, M.M., Hoekstra, A.Y., 2013. The water footprint of poultry, pork and beef: a comparative study in different countries and production systems. Water Resour. Ind. 1-2, 25–36. http://dx.doi.org/10.1016/j.wri.2013.03.001.

Gerten, D., Heinke, J., Hoff, H., Biemans, H., Fader, M., Waha, K., 2011. Global water availability and requirements for future food production. J. Hydrometeorol. 12 (5), 885–899. http://dx.doi.org/10.1175/2011JHM1328.1.

González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L., 2012. Comparative environmental performance of three different annual energy crops for biogas production in northern Italy. J. Clean. Prod. 43, 71–83.

Hastings, E., Pegram, G., 2012. Literature review for the applicability of water footprints in South Africa. WRC Report No. 2099/P/11. Water Research Commission, Gezina, South Africa. ISBN: 978-1-4312-0237-9.

Herath, I., Green, S., Horne, D., Singh, R., Clothier, B., 2014. Quantifying and reducing the water footprint of rain-fed potato production, part I: measuring the net use of blue and green water. J. Clean. Prod. 81, 111–119. http://dx.doi.org/10.1016/j.jclepro.2014.06.026.

Hess, T.M., Lennard, A.T., Daccache, A., 2015. Comparing local and global water scarcity information in determining the water scarcity footprint of potato cultivation in Great Britain. J. Clean. Prod. 87, 666–674. http://dx.doi.org/10.1016/j.jclepro.2014.10.075.

Hoekstra, A.Y., 2003. Virtual water trade. Proceedings of the international expert meeting on virtual water trade. International Expert Meeting on Virtual Water Trade 12(12), pp. 1–244.

Hoekstra, A.Y., 2010. The water footprint: water in the supply chain. Water 93, 12–13.

Hoekstra, A.Y., 2013. Sustainable, efficient, and equitable water use: the three pillars under wise freshwater allocation. Wiley Interdiscip. Rev. Water 1, 31–40. http://dx.doi.org/10.1002/wat2.1000.

Hoekstra, A.Y., Chapagain, A.K., 2006. The water footprints of Morocco and the Netherlands. Water Research Report Series No. 21. UNESCO-IHE, Delft, The Netherlands.

Hoekstra, A.Y., Chapagain, A.K., 2008. Globalization of Water. Sharing the Planet's Freshwater Resources. Blackwell Publishing.

Hoekstra, A.Y., Hung, P.Q., 2002. Virtual water trade. A quantification of virtual water flows between nations in relation to international trade. International Expert Meeting on Virtual Water Trade 12(11), pp. 1–244.

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. The Water Footprint Assessment Manual. (February 2011, http://doi.org/978-1-84971-279-8).

Hoekstra, A.Y., Booij, M.J., Hunink, J.C., Meijer, K.S., 2012. Blue water footprint of agriculture, industry, households and water management in the Netherlands. Water Research Report Series No. 58. UNESCO-IHE, Delft, The Netherlands.

Hoff, H., Falkenmark, M., Gerten, D., Gordon, L., Karlberg, L., Rockström, J., 2010. Greening the global water system. J. Hydrol. 384 (3–4), 177–186. http://dx.doi.org/10.1016/j.jhydrol.2009.06.026.

Huang, J., Zhang, H., Tong, W., Chen, F., 2012. The impact of local crops consumption on the water resources in Beijing. J. Clean. Prod. 21 (1), 45–50. http://dx.doi.org/10.1016/j.jclepro.2011.09.014.

Ingrao, C., Matarazzo, A., Tricase, C., Clasadonte, M.T., Huisingh, D., 2015a. Life cycle assessment for highlighting environmental hotspots in Sicilian peach production systems. J. Clean. Prod. 92, 109–120.

Ingrao, C., Tricase, C., Cholewa-wójcik, A., Kawecka, A., Rana, R., Siracusa, V., 2015b. Polylactic acid trays for fresh-food packaging: a Carbon Footprint assessment. Sci. Total Environ. 537, 385–398. http://dx.doi.org/10.1016/j.scitotenv.2015.08.023.

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.

ISO 14040 series, 2006. Environmental Management — Life Cycle Assessment — Principles and Framework.

ISO 14046, 2014. Environmental Management — Water Footprint — Principles, Requirements and Guidelines.

Jefferies, D., Muñoz, I., Hodges, J., King, V.J., Aldaya, M., Ercin, A.E., i Canals, L.M., Hoekstra, A.Y., 2012. Water footprint and life cycle assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. J. Clean. Prod. 33, 155–166. http://dx.doi.org/10.1016/j.jclepro.2012.04.015.

King, C.W., Webber, M.E., 2008. Water intensity of transportation. Environ. Sci. Technol. 42 (21), 7866–7872. http://dx.doi.org/10.1021/es800367m.

Konar, M., Hussein, Z., Hanasaki, N., Mauzerall, D.L., Rodriguez-Iturbe, I., 2013. Virtual water trade flows and savings under climate change. Hydrol. Earth Syst. Sci. 17 (8), 3219–3234. http://dx.doi.org/10.5194/hess-17-3219-2013.

Laane, R.W.P.M., Brockmann, U., van Liere, L., Bovelander, R., 2005. Immission targets for nutrients (N and P) in catchments and coastal zones: a North Sea assessment. Estuar. Coast. Shelf Sci. 62 (3), 495–505.

Lamastra, L., Suciu, N.A., Novelli, E., Trevisan, M., 2014. A new approach to assessing the water footprint of wine: an Italian case study. Sci. Total Environ. 490, 748–756. http://dx.doi.org/10.1016/j.scitotenv.2014.05.063.

Lee, Y.-J., 2015. Land, carbon and water footprints in Taiwan. Environ. Impact Assess. Rev. 54, 1–8. http://dx.doi.org/10.1016/j.eiar.2015.04.004.

Liu, C., Kroeze, C., Hoekstra, A.Y., Gerbens-Leenes, P.W., 2012. Past and future trends in grey water footprints of anthropogenic nitrogen and phosphorus inputs to major world rivers. Ecol. Indic. 18, 42–49. http://dx.doi.org/10.1016/j.ecolind.2011.10.005.

Liu, J., Sun, S., Wu, P., Wang, Y., Zhao, X., 2015. Evaluation of crop production, trade, and consumption from the perspective of water resources: a case study of the Hetao irrigation district, China, for 1960–2010. Sci. Total Environ. 505, 1174–1181. http://dx.doi.org/10.1016/j.scitotenv.2014.10.088.

Manzardo, A., Mazzi, A., Loss, A., Butler, M., Williamson, A., Scipioni, A., 2015. Lessons learned from the application of different water footprint approaches to compare different food packaging alternatives. J. Clean. Prod. http://dx.doi.org/10.1016/j.jclepro.2015.08.019.

Marano, R.P., Filippi, R.A., 2015. Water Footprint in paddy rice systems. Its determination in the provinces of Santa Fe and Entre Ríos, Argentina. Ecol. Indic. 56, 229–236. http://dx.doi.org/10.1016/j.ecolind.2015.03.027.

Mekonnen, M.M., 2011. Spatially and Temporally Explicit Water Footprint Accounting (PhD Thesis) http://dx.doi.org/10.3990/1.9789036532211.

Mekonnen, M.M., Hoekstra, A.Y., 2010. A global and high-resolution assessment of the green, blue and grey water footprint of wheat. Hydrol. Earth Syst. Sci. 14 (7), 1259–1276. http://dx.doi.org/10.5194/hess-14-1259-2010.

Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrol. Earth Syst. Sci. 15, 1577–1600.

Mekonnen, M.M., Hoekstra, A.Y., 2012. A global assessment of the water footprint of farm animal products. Ecosystems 15 (3), 401–415. http://dx.doi.org/10.1007/s10021-011-9517-8.

Mekonnen, M.M., Hoekstra, A.Y., 2014. Water footprint benchmarks for crop production: a first global assessment. Ecol. Indic. 46, 214–223. http://dx.doi.org/10.1016/j.ecolind.2014.06.013.

Mekonnen, M.M., Hoekstra, A.Y., 2015. Global grey water footprint and water pollution levels related to anthropogenic nitrogen loads to fresh water. Environ. Sci. Technol. http://dx.doi.org/10.1021/acs.est.5b03191 (151006154421000).

Mekonnen, M., Pahlow, M., Aldaya, M., Zarate, E., Hoekstra, A.Y., 2015a. Sustainability, efficiency and equitability of water consumption and pollution in Latin America and the Caribbean. Sustainability 7 (2), 2086–2112. http://dx.doi.org/10.3390/su7022086.

Mekonnen, M.M., Gerbens-Leenes, P.W., Hoekstra, A.Y., 2015b. The consumptive water footprint of electricity and heat: a global assessment. Environ. Sci. Water Res. Technol. http://dx.doi.org/10.1039/C5EW00026B.

Mordini, M., Nemecek, D.T., Gaillard, D.G., 2009. Carbon and water footprint of oranges and strawberries a literature review. Agroscope Reckenholz-Tanikon Research Station, Switzerland.

Nana, E., Corbari, C., Bocchiola, D., 2014. A model for crop yield and water footprint assessment: study of maize in the Po valley. Agric. Syst. 127, 139–149. http://dx.doi.org/10.1016/j.agsy.2014.03.006.

Natali, F., Marta, A.D., Orlando, F., Orlandini, S., 2009. Water use in Italian agriculture: analysis of rainfall patterns. Ital. J. Agrometeorol. 55 (3), 56–59.

Nelson, G.C., Valin, H., Sands, R.D., Havlík, P., Ahammad, H., Deryng, D., ... Willenbockel, D., 2014. Climate change effects on agriculture: economic responses to biophysical shocks. Proc. Natl. Acad. Sci. U. S. A. 111 (9), 3274–3279. http://dx.doi.org/10.1073/pnas.1222465110.

Oki, T., Sato, M., Kawamura, A., 2003. Virtual water trade to Japan and in the world. Virtual Water Trade, pp. 1–13.

Orlowsky, B., Hoekstra, A.Y., Gudmundsson, L., Seneviratne, S.I., 2014. Today's virtual water consumption and trade under future water scarcity. Environ. Res. Lett. 9 (7), 074007. http://dx.doi.org/10.1088/1748-9326/9/7/074007.

Page, G., Ridoutt, B., Bellotti, B., 2012. Carbon and water footprint tradeoffs in fresh tomato production. J. Clean. Prod. 32, 219–226. http://dx.doi.org/10.1016/j.jclepro.2012.03.036.

Pahlow, M., Van Oel, P.R., Mekonnen, M.M., Hoekstra, A.Y., 2015. Increasing pressure on freshwater resources due to terrestrial feed ingredients for aquaculture production. Sci. Total Environ. 536, 847–857. http://dx.doi.org/10.1016/j.scitotenv.2015.07.124.

Palhares, J.C.P., Pezzopane, J.R.M., 2015. Water footprint accounting and scarcity indicators of conventional and organic dairy production systems. J. Clean. Prod. http://dx.doi.org/10.1016/j.jclepro.2015.01.035.

Pellegrini, G., Ingrao, C., Camposeo, S., Tricase, C., 2015. Application of Water Footprint to olive growing systems in the Apulia region: a comparative assessment. J. Clean. Prod. 2015. http://dx.doi.org/10.1016/j.jclepro.2015.10.088.

Pfister, S., Bayer, P., 2014. Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. J. Clean. Prod. 73, 52–62. http://dx.doi.org/10.1016/j.jclepro.2013.11.031.

Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43 (11), 4098–4104. http://dx.doi.org/10.1021/es802423e.

Pfister, S., Bayer, P., Koehler, A., Hellweg, S., 2011. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environ. Sci. Technol. 45 (13), 5761–5768. http://dx.doi.org/10.1021/es1041755.

Ramírez, T., Meas, Y., Dannehl, D., Schuch, I., Miranda, L., Rocksch, T., Schmidt, U., 2015. Water and carbon footprint improvement for dried tomato value chain. J. Clean. Prod. 104, 98–108. http://dx.doi.org/10.1016/j.jclepro.2015.05.007.

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob. Environ. Chang. 20 (1), 113–120. http://dx.doi.org/10.1016/j.gloenvcha.2009.08.003.

Rinaldi, S., Barbanera, M., Lascaro, E., 2014. Assessment of carbon footprint and energy performance of the extra virgin olive oil chain in Umbria, Italy. Sci. Total Environ. 482-483, 71–79. http://dx.doi.org/10.1016/j.scitotenv.2014.02.104.

Rodriguez, C.I., Ruiz de Galarreta, V.A., Kruse, E.E., 2015. Analysis of water footprint of potato production in the pampean region of Argentina. J. Clean. Prod. 90, 91–96. http://dx.doi.org/10.1016/j.jclepro.2014.11.075.

Roelich, K., Owen, A., Thompson, D., Dawkins, E., West, C., 2014. Improving the policy application of footprint indicators to support Europe's transition to a one planet economy: the development of the EUREAPA tool. Sci. Total Environ. 481, 662–667. http://dx.doi.org/10.1016/j.scitotenv.2013.10.084.

Rosegrant, M.W., Cai, X., Cline, S.A., 2007. Averting an impending crisis. N. Engl. J. Public Policy 21.

Ruini, L., Marino, M., Pignatelli, S., Laio, F., Ridolfi, L., 2013. Water footprint of a large-sized food company: the case of Barilla pasta production. Water Resour. Ind. 1-2, 7–24. http://dx.doi.org/10.1016/j.wri.2013.04.002.

Scarpare, F.V., Hernandes, T.A.D., Ruiz-Corrêa, S.T., Kolln, O.T., Gava, G.J.D.C., dos Santos, L.N.S., Victoria, R.L., 2016. Sugarcane water footprint under different management practices in Brazil: Tietê/Jacaré watershed assessment. J. Clean. Prod. 112, 4576–4584. http://dx.doi.org/10.1016/j.jclepro.2015.05.107.

Schyns, J.F., Hoekstra, A.Y., 2014. The added value of Water Footprint Assessment for national water policy: a case study for Morocco. PLoS One 9 (6). http://dx.doi.org/10.1371/journal.pone.0099705.

Song, G., Li, M., Semakula, H.M., Zhang, S., 2015. Food consumption and waste and the embedded carbon, water and ecological footprints of households in China. Sci. Total Environ. 529, 191–197. http://dx.doi.org/10.1016/j.scitotenv.2015.05.068.

Steen-Olsen, K., Weinzettel, J., Cranston, G., Ercin, A.E., Hertwich, E.G., 2012. Carbon, land, and water footprint accounts for the European Union: consumption, production, and displacements through international trade. Environ. Sci. Technol. 46 (20), 10883–10891. http://dx.doi.org/10.1021/es301949t.

Su, M.-H., Huang, C.-H., Li, W.-Y., Tso, C.-T., Lur, H.-S., 2015. Water footprint analysis of bioethanol energy crops in Taiwan. J. Clean. Prod. 88, 132–138. http://dx.doi.org/10.1016/j.jclepro.2014.06.020.

Sun, S., Wu, P., Wang, Y., Zhao, X., Liu, J., Zhang, X., 2013. The impacts of interannual climate variability and agricultural inputs on water footprint of crop production in an irrigation district of China. Sci. Total Environ. 444, 498–507. http://dx.doi.org/10.1016/j.scitotenv.2012.12.016.

Thaler, S., Zessner, M., Bertran De Lis, F., Kreuzinger, N., Fehringer, R., 2012. Considerations on methodological challenges for water footprint calculations. Water Sci. Technol. 65, 1258–1264. http://dx.doi.org/10.2166/wst.2012.006.

Tran, T., Da, G., Moreno-Santander, M.A., Vélez-Hernández, G.A., Giraldo-Toro, A., Piyachomkwan, K., ... Dufour, D., 2015. A comparison of energy use, water use and carbon footprint of cassava starch production in Thailand, Vietnam and Colombia. Resour. Conserv. Recycl. 100, 31–40. http://dx.doi.org/10.1016/j.resconrec.2015.04.007.

US-Environmental Protection Agency, 2005. http://www.epa.gov (Accessed 27 September 2015).

Van Oel, P.R., Hoekstra, A.Y., 2012. Towards quantification of the water footprint of paper: a first estimate of its consumptive component. Water Resour. Manag. 26 (3), 733–749. http://dx.doi.org/10.1007/s11269-011-9942-7.

Vanham, D., 2015. Does the water footprint concept provide relevant information to address the water–food–energy–ecosystem nexus? Ecosyst. Serv. 1–10. http://dx.doi.org/10.1016/j.ecoser.2015.08.003.

Vanham, D., Bidoglio, G., 2013. A review on the indicator water footprint for the EU28. Ecol. Indic. 26 (2013), 61–75. http://dx.doi.org/10.1016/j.ecolind.2012.10.021.

Vanham, D., Mekonnen, M.M., Hoekstra, A.Y., 2013. The water footprint of the EU for different diets. Ecol. Indic. 32, 1–8. http://dx.doi.org/10.1016/j.ecolind.2013.02.020.

Vörösmarty, C.J., Green, P., Salisbury, J., Lammers, R.B., 2000. Global water resources: vulnerability from climate change and population growth. Science (New York, N.Y.) 289 (5477), 284–288. http://dx.doi.org/10.1126/science.289.5477.284.

Wang, Y.B., Wu, P.T., Engel, B.a., Sun, S.K., 2014. Application of water footprint combined with a unified virtual crop pattern to evaluate crop water productivity in grain production in China. Sci. Total Environ. 497-498, 1–9. http://dx.doi.org/10.1016/j.scitotenv.2014.07.089.

Wang, X., Li, X., Fischer, G., Sun, L., Tan, M., Xin, L., Liang, Z., 2015. Impact of the changing area sown to winter wheat on crop water footprint in the North China Plain. Ecol. Indic. 57, 100–109. http://dx.doi.org/10.1016/j.ecolind.2015.04.023.

Watkins, K., Carvajal, L., Coppard, D., Fuentes, R., Ghosh, A., Giamberardini, C., Johansson, C., Seck, P., Ugaz, C., Yaqub, S., 2006. United Nations Development Program, Human Development Report. Macmillan 0-230-50058-7.

Wichelns, D., 2015. Virtual water and water footprints do not provide helpful insight regarding international trade or water scarcity. Ecol. Indic. 52, 277–283. http://dx.doi.org/10.1016/j.ecolind.2014.12.013.

Xu, Y., Huang, K., Yu, Y., Wang, X., 2015. Changes in water footprint of crop production in Beijing from 1978 to 2012: a logarithmic mean Divisia index decomposition analysis. J. Clean. Prod. 87, 180–187. http://dx.doi.org/10.1016/j.jclepro.2014.08.103.

Yang, H., Wang, L., Abbaspour, K.C., Zehnder, A.J.B., 2006. Virtual water trade: an assessment of water use efficiency in the international food trade. Hydrol. Earth Syst. Sci. 10 (3), 443–454. http://dx.doi.org/10.5194/hess-10-443-2006.

Yoo, S.-H., Choi, J.-Y., Lee, S.-H., Kim, T., 2013. Estimating water footprint of paddy rice in Korea. Paddy Water Environ. 12 (1), 43–54. http://dx.doi.org/10.1007/s10333-013-0358-2.

Zhang, G.P., Hoekstra, A.Y., Mathews, R.E., 2013. Water Footprint Assessment (WFA) for better water governance and sustainable development. Water Resour. Ind. 1-2, 1–6. http://dx.doi.org/10.1016/j.wri.2013.06.004.

Zhuo, L., Mekonnen, M.M., Hoekstra, A.Y., 2014. Sensitivity and uncertainty in crop water footprint accounting: a case study for the Yellow River basin. Hydrol. Earth Syst. Sci. 18 (6), 2219–2234. http://dx.doi.org/10.5194/hess-18-2219-2014.

Zimmer, D., Renault, D., 2003. Virtual water in food production and global trade: review of methodological issues and preliminary results. World Water Council. Virtual Water Trade 1, pp. 1–19.

Zoumides, C., Bruggeman, A., Hadjikakou, M., Zachariadis, T., 2014. Policy-relevant indicators for semi-arid nations: the water footprint of crop production and supply utilization of Cyprus. Ecol. Indic. 43, 205–214. http://dx.doi.org/10.1016/j.ecolind.2014.02.012.

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Beyond the Water Footprint: A new framework proposal to assess freshwater environmental impact and consumption

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ABSTRACT

Because the assessment of grey water according to the Water Footprint Network (WFN) permits to quantify the dilution volume to restore water quality considering the substance that demands the highest dilution volume, the effect of other polluting substances (e.g., nitrogen, phosphorous, pesticides) applied on field cannot be evaluated. Nevertheless, the environmental load of all these substances cannot be neglected, especially when huge amounts of organic fertilisers are spread.

Additionally, because blue water quantification with WFN permits to analyse only the water consumed by the crop (mainly for irrigation purposes), a method assessing the gross irrigation volume effectively applied on field was used (i.e. Water Footprint Applied - WFA).

A Pollution Water Indicator (PWI) was developed to denote the intensity of water pollution identifying the effect of the main polluting substances from crop cultivation. For PWI, both grey water and the water-related environmental impact categories (freshwater eutrophication, marine eutrophication and freshwater ecotoxicity) evaluated by means of Life Cycle Assessment (LCA) were considered.

In this context, this study proposes a framework for assessing both the environmental impact and the consumption of freshwater. Different organic fertilisers spreading techniques with different timing of incorporation and straw management and three irrigation technologies with variable technical efficiency were compared for WFA quantification of maize grain production in Northern Italy.

With regard to organic fertilisers spreading, PWI resulted better when nutrients leaching is reduced, while it was worse with fast soil incorporation and direct soil injection of organic fertilisers that, reducing ammonia volatilisation, involve higher nitrate losses. As concerns irrigation, sprinkler and drip irrigation are highly recommended because they permit to apply water volumes much close to the consumed ones, with blue water between −33% and −60% of total WFA with drip instead of surface irrigation.

1. Introduction

Current modes of production and consumption systems have been recognised as often unsustainable from an economic, social and environmental point of view (Blok et al., 2015). The production of food and feed is acknowledged to be responsible for huge Greenhouse Gas (GHG) emissions (Renzulli et al., 2015) and water depletion (Hess et al., 2016; Pellegrini et al., 2016). Therefore, several studies aiming at assessing and mitigating the environmental impact of agro-food (Nemecek et al., 2011) and agro-energy (González-García et al., 2012; Ingrao et al., 2015) productions have been realised along the years. Furthermore, several agricultural and industrial activities are closely intertwined with water consumption. The availability of sufficient freshwater resources constitutes a significant precondition for covering global consumer needs, in particular for products arising from the agro-food sector (Aivazidou et al., 2016). Freshwater consumption and pollution are the major environmental issues for which agriculture is responsible (Mekonnen and Hoekstra, 2011) and overconsumption is even increasing because of population growth and dietary requests. Nevertheless, it is fundamental to reduce the dependence of agricultural productions from finite natural resources in order to allow a fast transition to sustainable and equitable societies (Ingrao et al., 2016; Repar et al., 2016).

To quantify the volumetric freshwater consumption of agro-food products, the Water Footprint (WF) indicator (composed by

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green, blue and grey water from here on called WFgreen, WFblue and WFgrey, respectively) was developed (Hoekstra and Hung, 2002) and adopted by the Water Footprint Network (WFN).

The WF has gone through several methodological changes since its initial definition (Lovarelli et al., 2016a), especially to reduce the simplified calculation that had been developed. For instance, Jeswani and Azapagic (2011) reviewed the methods available to quantify water use and environmental impacts of freshwater, documenting variability in the results due to the different methods used (e.g., Milà I Canals et al., 2009; Ridoutt and Pfister, 2010) and the lack of methodological consistency. Lamastra et al. (2014) compared the results of the WFgrey (water volume to restore water quality) with their method for water quality restoration affected by pesticides emissions and highlighted not negligible differences between the two approaches as well. Several authors (Bayart et al., 2010; Jefferies et al., 2012; Manzardo et al., 2016; Pfister et al., 2009) underlined also the need of analysing the consumptive water use instead of the volumetric one, quantifying the WF in compliance with the ISO 14046 (ISO 14046, 2014).

Among the three WF components, WFgrey highlights the major problems, and studies aimed to develop indicators for better describing freshwater quality degradation and consumptive use. For instance, Zonderland-Thomassen et al. (2014), Herath et al. (2013) and Bayart et al. (2014) introduced specific indicators and indexes and combined their evaluation with WFgrey. In particular, in those studies as well as in WFN, WFgrey was quantified as the dilution volume of the substance that demands for the highest volume to improve water quality, although authors documented the usefulness of including also the assessment of other minor pollutants. Accordingly, even if the dilution permits to respect the normative threshold for freshwater quality, all polluting substances affect the system (Wu et al., 2016) and stockpile along time. Therefore, their environmental effect is not null and should be studied to promote and develop less detrimental productions.

Additionally, critics on the WFN method concern also WFblue, defined as the evapotranspirated water that derives from human intervention on ground and surface water and that does not return to the water system (Hoekstra, 2010). Several authors (Scarpare et al., 2015; Yoo et al., 2013) did not strictly follow the WFN approach and they considered the effective irrigation water volume applied on field; for the same reason, García Morillo et al. (2015) developed the Water Footprint Applied (WFA) method. The main issues they raised are that irrigation water in WFblue lacks in considering the effective volume applied at the technical gross efficiency of the irrigation method and that this can represent a great problem where water availability is a limiting factor. In particular, reducing the effective volume applied would entail decreasing the dependence on the natural resources, both for water withdrawal and for fossil fuel consumption during pumping, in view of improving a long-term efficient system.

In this context, the study was aimed to develop a new assessment framework that considers, for freshwater, both the environmental impact and the consumption. To these aims two main activities were carried out:

(i) developing a Pollution Water Indicator (PWI) to denote the intensity of water pollution due to all main pollutants that affect the agricultural productive system and to give a better picture of water pollution. In particular, PWI identifies the effect of nutrients and pesticides losses on water pollution;
(ii) quantifying the effectively used water for irrigation with the Water Footprint Applied (WFA) method instead of the consumed (WFN) one.

To study both pollution and use of freshwater, maize grain production was identified as case study, mainly because maize cultivation is very common in Po Valley and high yield can be reached thanks to both climate and availability of production inputs (e.g., fertilisers and water). As regard to fertilisers, North Italy is highly devoted to animal breeding and huge amounts of organic fertilisers need to be managed (Bacenetti et al., 2016b). Moreover, there is a huge availability of freshwater (Lovarelli et al., 2016a), supported by a complex irrigation network that justifies why freshwater has never been considered as a scarce issue to deal with. Nevertheless, agriculture is facing water reductions for irrigation purposes and high-efficient technologies must spread. In this context, this study can be helpful for environmental indicators practitioners, and in particular for freshwater indicators practitioners, in order to widely analyse freshwater environmental concerns and respond to the challenges of a bio-based economy.

Finally, it should be observed that this study was designed to provide answers to the following questions:

(i) are there any consequences on freshwater resource and on Water Footprint when different techniques for the application of nutrients on field are considered and different irrigation techniques are adopted?
(ii) can a single index improve the actual WFgrey quantification to describe the environmental impact of freshwater consumption and quality degradation?

The manuscript has been organised as follows: in Section 2 the theoretical approach for the development and the assessment of the PWI is presented; in Section 3 the case study for the application of PWI is described; in Section 4 the case study results are presented and in Section 5 the application of PWI is discussed.

2. Theoretical model

Fig. 1 shows the methodological framework of the study.

PWI has been built taking into account four sub-indicators. They include:

(i) grey water, assessed with the Water Footprint Network (WFN) method; and
(ii) three impact categories (i.e. Freshwater Eutrophication, Marine Eutrophication and Freshwater Ecotoxicity), assessed with the Life Cycle Assessment (LCA) approach.

Grey water was selected in order to consider the potential volume necessary to restore water quality due to the substance requiring the highest dilution volume. The three evaluated impact categories, instead, were selected to consider the potential environmental impact of all the main pollutants (e.g., nutrients and pesticides) that are released to water within the analysed system boundary and that can impact on freshwater resource. By developing the PWI it is possible, therefore, to comprehensively identify a trade-off between the potential volume used to restore water quality due to the most polluting substance and the environmental load caused by pollutants emitted to water.

The Pollution Water Indicator (PWI) is modelled in a graph developed considering WFgrey, FE, ME and FEx as the vertexes of a rhombus whose area represents the PWI: the smaller the area, the lower the freshwater pollution. In addition, since the attribution of different weights to WFgrey and to the 3 impact categories is subjective, for the time being, they have been considered having the same weight on the final score.

As concerns the WF quantification, both the WFN and WFA approaches (García Morillo et al., 2015) were used. In more details, the WFN was the identified method for quantifying green and grey

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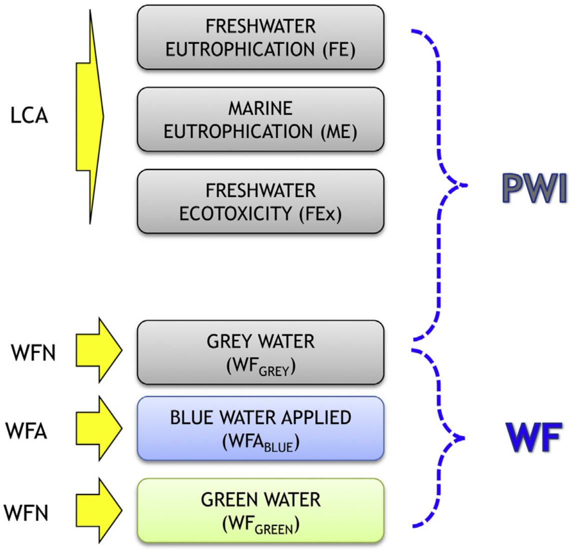

Fig. 1. The environmental impact on freshwater is assessed quantifying: (i) the Pollution Water Indicator (PWI) considering Freshwater Eutrophication, Marine Eutrophication and Freshwater Ecotoxicity evaluated by means of the Life Cycle Assessment (LCA) method and the WFgrey with the Water Footprint Network (WFN); (ii) the WF considering the WFN for grey and green water and the Water Footprint Applied (WFA) for blue water.

water, whereas blue water was quantified with the WFA one, in order to calculate the irrigation volume applied (i.e. pumped water on field) instead of the crop used (evapotranspirated) water.

2.1. *Water Footprint assessment*

The Water Footprint Network (WFN) quantifies 3 components of water (WFgreen, WFblue and WFgrey). In this study, its methodology was followed for WFgreen (evapotranspirated water from precipitation) and WFgrey (dilution volume to assimilate pollutants load and restore water quality) components (Hoekstra, 2010), following the approach described in the Supplementary Material.

On the contrary, blue water was assessed considering the irrigation efficiency of the adopted technique (i.e. surface irrigation, sprinkler irrigation and drip irrigation) and quantifying the water used in the system according to the Water Footprint Applied (WFA) method and the Relative Supply Indicator (RIS)[1] (García Morillo et al., 2015). The reason is the necessity of considering the local water availability for irrigation supply (at gross technical efficiency): the effective irrigation volume must be considered because it is the gross volume that must be applied, although only one part is effectively consumed by the crop (evapotranspirated). In fact, irrigation losses affect the amount of irrigation water to apply; if this volume is not available, the net water (consumed by the crop) cannot be achieved and the crop's growth needs are not satisfied. The less efficient the irrigation method, the higher water volume must be pumped in.

Therefore, the two methods for WF assessment, WFN and WFA, are equal for the quantification of WFgreen and WFgrey but differ for WFblue that, in WFA, is computed considering the efficiency of the irrigation system.

2.2. *Life Cycle Assessment*

The Life Cycle Assessment (LCA) is a fundamental method to quantify the environmental impact for which the studied system is responsible and it is performed following the ISO Standards (ISO 14044, 2006).

LCA has been significantly improved along time and so has become more systematic and robust for the identification and quantification of the potential impacts and improvements associated with a product in its life cycle. This methodology is increasingly being applied as to investigate several fields like, for instance, agriculture, thereby becoming an invaluable decision-support tool for farmers, researchers, policy makers and other stakeholders (Ingrao et al., 2015).

In this study, the goal is to quantify the environmental impact on freshwater resource for maize grain cultivation on a selected farm.

In accordance with previous studies where a mass based functional unit (FU) was selected (Bacenetti et al., 2015a, 2016a; Fedele et al., 2014; Renzulli et al., 2015), in this study the selected FU is 1 ton of maize grain at 14% commercial moisture.

The system boundary includes inputs and outputs for maize cultivation (Fig. 2). A cradle to farm gate perspective was adopted, which means that the included inputs and outputs are: raw materials extraction (e.g., fossil fuels), manufacture of agricultural inputs (e.g., seed, fertilisers, pesticides, tractors and implements) and energy, supply and use of agricultural inputs (emissions of fertilisers, pesticides, diesel fuel and tire abrasion), maintenance and final disposal of machines.

For the Life Cycle Impact Assessment (LCIA) phase, the composite method recommended by the International Reference Life Cycle Data System (ILCD) (Wolf et al., 2012) is used. The following impact categories affecting the freshwater system are chosen: freshwater eutrophication (FE; kg P eq), marine eutrophication (ME; kg N eq) and freshwater ecotoxicity (FEx CTUeq). Their environmental impact is due to the emissions from nutrients application ($PO4^-$ emissions into water and NO_3—N leaching for FE and ME, respectively) and from pesticides application (considering the active ingredients into the environment for FEx).

3. Materials and methods

3.1. *Goal and scope definition and functional unit selection*

All the four sub-indicators (WFgrey, FE, ME and FEx) used to assess the PWI as well as WFgreen and WFAblue that, together with WFgrey, are used to compute the WFA, are referred to the same functional unit (FU) (1 ton of maize grain at 14% moisture).

3.2. *System description*

The studied farm is located in the District of Milan, in the Italian Po Valley. The local geographic coordinates are 45°11′31″ N and 9°28′35″ E and the District is on average at 104 m above the sea level. Climate is characterised by mild winter and dry hot summer, with rainfall concentrated in spring and autumn. Soil texture is of medium texture (sandy-loamy). Since irrigation water is locally available, the study refers to irrigated field working conditions. The farm agricultural land area is 40 ha, of which 25 ha are cultivated with maize grain. As largely occurs in Po Valley (Negri et al., 2014), the cultivated crop is maize grain FAO Class 700 in a single cropping system, characterised by a long growing period (120–160 days). Maize cultivation practice is characterised by mechanical operations, working features and yields often year by year similar. Therefore, the cultivation method in this study is comparable to other studies (Bacenetti et al., 2015b, 2016b; Negri et al., 2014; Noya et al., 2015).

The organic fertiliser (85 t/ha of pig slurry, characterised by a nutrient content of $N = 0.24\%$, $P_2O_5 = 0.25$, $K_2O = 0.55$) is spread

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[1] Indicator that informs on how the irrigation volume applied matches the theoretical water requirements of the growing season (García Morillo et al., 2015).

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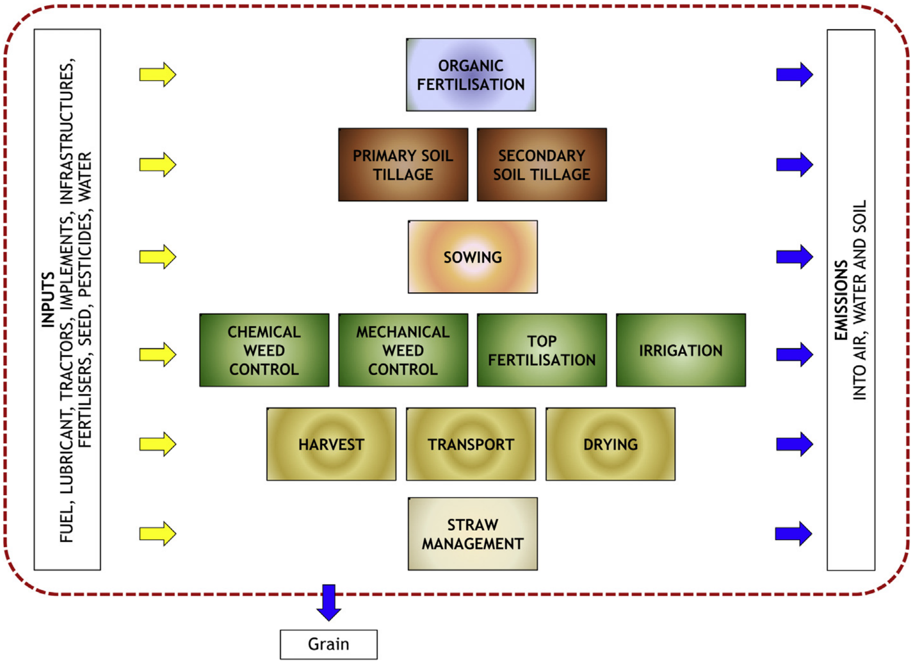

Fig. 2. System boundary of maize cultivation.

with a superficial spreading technique during spring; a slurry tanker of 20 m^3 is coupled with a 4WD (drive wheels) 130-kW tractor and the incorporation is carried out by means of ploughing after more than 72 h from the spreading. Primary and secondary soil tillage are carried out with a 3-furrows mouldboard plough and a rotary harrow, both coupled with a 4WD 90-kW tractor. Seeds are sown with a precision seeder coupled with a 4WD 62-kW tractor. As top fertilisation, 60 kg/ha of urea are spread with a mineral fertiliser spreader coupled with the 4WD 62-kW tractor. Surface irrigation is also carried out with the same tractor (4WD; 62 kW) coupled with a pump and following a turn schedule equal to 7 days fixed by a local Consortium. Surface irrigation is a traditional technique (Giardini, 2003) characterised by low irrigation efficiency (50%). Although it is technically outdated in several parts of the world, it is still much practised in some areas of Po Valley thanks to the high water availability. In each intervention about 140 mm water are furnished. Irrigation is not accounted during the 30 days before the harvest, in order to support maize ripening.

As concerns straw management, it is chopped with a straw chopper coupled with the 4WD 90-kW tractor after grain harvest and is left on field during winter. Grain is dried and stored on farm.

3.2.1. Alternatives to the baseline practice

Some of the field operations for maize grain cultivation that considerably affect freshwater degradation and consumption have been investigated as alternatives to the baseline practice (BS). These operations refer to:

1) Fertiliser spreading and straw management

The spreading techniques and the straw management can affect the amount of nutrients applied to the soil as well as the amount removed with crop harvesting. Surface spreading with incorporation (after different timings) and direct injection are two techniques of growing interest on the environmental point of view, especially for their potential beneficial effect on terrestrial acidification and eutrophication (Bacenetti et al., 2016b). With this regard, the following alternatives to the baseline practice for organic fertiliser spreading (i.e. surface slurry spreading incorporated after >72 h) have been assessed:

(i) “fast incorporation” (S1), with incorporation of pig slurry within 2 h after spreading and the same mechanisation as in BS;
(ii) “soil injection” (S2), with direct injection of pig slurry in the soil in 7–10 cm deep furrows. A slurry tanker (20 m^3) is equipped with 5 anchors and is coupled to a 4WD 180-kW tractor;
(iii) “straw collection” (S3), in which the difference from BS arises from the collection and baling of straw, involving a higher nutrient removal and a consequent higher organic fertiliser mass applied;
(iv) “digestate” (S4), in which the same mechanical features as BS are considered, but instead of pig slurry, digestate from a biogas plant fed with maize silage and pig slurry is used;
(v) “only mineral” (S5), in which no organic fertiliser is spread but urea (500 kg/ha) and triple superphosphate (150 kg/ha).

Table 1
Irrigation efficiency, intervention threshold and volume for each of the irrigation methods assessed.

Irrigation method	Acronym	Irrigation efficiency [a] (η_{irr}; %)	Intervention threshold [b] (β; %)	Irrigation volume per interv [c] (V; mm)
Surface irrigation	SI	50%	90%	140
Sprinkler irrigation	PI	70%	65%	50
Drip irrigation	DI	90%	60%	10

[a] ERSAF, 2016.
[b] Giardini, 2003.
[c] Allen et al., 1998.

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Table 2
Climate data for the 5 years considered.

Climate parameters	Unit	Year				
		2011	2012	2013	2014	2015
Rainfall	mm	198.8	611.0	848.2	1116.2	232.0
Average temperature, min	°C	9.8	9.8	9.7	10.8	10.5
Average temperature, max	°C	18.4	18.8	17.9	19.0	19.3
Average relative humidity, min	%	54.9%	56.0%	64.4%	63.9%	63.0%
Average relative humidity, max	%	89.7%	91.4%	95.5%	97.1%	96.0%
Average global solar radiation	W/m^2	165.1	165.9	155.1	156.4	166.1
Average wind speed	m/s	1.0	1.9	2.8	2.8	4.6

In order to apply the same amount of nitrogen for the crop nutrients demand, in S1 and S2, 85 t/ha of pig slurry were applied, in S3 123 t/ha of pig slurry were distributed (in order to respond to the higher nutrient removal due to the straw collection) and in S4, the applied digestate was 56.4 t/ha, since digestate has a higher mineral fertiliser equivalency (MFE) than pig slurry (Lijó et al., 2014). During top fertilisation, in S1, S2 and S4 60 kg/ha of mineral fertiliser (urea) were applied, while in S3 urea was 86 kg/ha, since the same nitrogen proportion used in BS was applied.

2) Irrigation techniques

The different irrigation methods are characterised by different efficiency and, consequently, considering the water needed as a constant, to different water volumes. To surface irrigation (SI), two alternative techniques were studied:

(i) “sprinkler irrigation with hose reel” (PI), characterised by higher irrigation efficiency[2] than SI (70%) is spreading in several areas of Po Valley. It is commonly carried out with a mobile hose reel on field and involves about 50 mm water applied per intervention;
(ii) “drip irrigation” (DI), it is the most efficient irrigation technology (efficiency 90%) and its application on maize fields is spreading but still has a limited use. In Italy, less than 5% maize fields are irrigated through DI (ISTAT, 2016). Per each intervention, not affected by irrigation turns because they use well water systems, about 10 mm water are furnished.

In PI, similarly to SI, the turn irrigation obliges to focus on the weather conditions in order to avoid water deficit and extreme water stress. Because of the irrigation turn, intervention threshold[3] (β; %) before the excessive water stress must be lower than in DI. This last, in fact, is characterised by an automatic irrigation system that pumps well water on field: when the crop water availability decreases below the defined threshold, the farmer can irrigate thanks to well water.

Table 1 shows the operative parameters for the evaluated irrigation methods.

3.3. Inventory data collection

Grain yield from 2011 to 2015 as well as information about the maize cultivation system adopted, mechanical operations and farm work organisation were gathered from interviews with the farmer. Yield ranged between 9.75 t/ha and 14.50 t/ha, being on average 12.3 t/ha at 14% commercial moisture. Meteorological data were collected daily, on the same timeframe, from the closest station to the farm (Regional Institution for Protection of the Environment ARPA, 2016). These data are reported as yearly average in Table 2. In more details, were gathered: rainfall (P; mm), minimum and maximum temperature (T_{min} and T_{max}; °C), minimum and maximum relative humidity (RH_{min} and RH_{max}; %), global solar radiation (G; W/m^2) and wind speed (W; m/s).

With regard to the cultivation schedule, maize grain was sown in April (3rd April 2011, 1st April 2012, 12th April 2013, 26th April 2014 and 15th April 2015). The harvesting was carried out in September (4th in 2011, 8th in 2012, 19th in 2013, 14st in 2014 and 21st in 2015), with crop growing cycles equal, on average, to 158 days. Phenological phases, root depth, average Leaf Area Index and evapotranspiration coefficient (kc) were obtained from FAO Papers n. 56 (Allen et al., 1998) and are reported in Table S1 (Supplementary Material). These data were used to calculate crop evapotranspiration and hydrological water balance for quantifying WFgreen and WFAblue in accordance with Allen et al. (1998) and as described in the Supplementary Material (Sections S1 and S2).

Data on farm mechanisation are reported in Table 3. Mass of materials composing tractors and implements that is depleted per each operation along the machinery lifespan, fuel consumed and exhaust gases emitted were quantified with the model ENVIAM (ENVironmental Inventory of Agricultural Machinery operations) (Lovarelli et al., 2016b) according to the primary data (e.g., type of machinery, working time per operation) furnished at the interview. Background data necessary for the Life Cycle Inventory phase of LCA derive from Ecoinvent Database v3 (Althaus et al., 2007; Frischknecht et al., 2007; Jungbluth et al., 2007; Nemecek and Kägi, 2007; Spielmann et al., 2007) and are reported in Table S2 (Supplementary Material).

No environmental load was considered for pig slurry, because it is a livestock system waste (González-García et al., 2012; Niero et al., 2015).

No allocation was performed in any alternative solution except for S3, where an economic allocation between grain and straw was assessed. As average of 2011–2015, yield was 12.3 and 14.5 t/ha for grain and straw, respectively, and market price was 180 and 35 €/ha, respectively (Milan Grain Association, 2016); this determines an allocation equal to 82% and 18%, respectively for grain and straw. Yield per year is reported in Table S3 (Supplementary Material).

The NO_3–N leaching fractions were calculated with the model EFE-So (Estimation of Fertilisers Emissions-Software, available at: http://www.sustainable-systems.org.uk/tools.php) (Fusi and Bacenetti, 2014) based on Brentrup et al. (2000). As concerns phosphorous, leaching and runoff were quantified for phosphate (PO_4^-) following Prahsun (2006) and Nemecek and Kägi (2007). Accordingly, as factors for phosphorous emissions into water were considered: 0.07 kg $P \cdot ha^{-1} \cdot year^{-1}$ for leaching to ground water and 0.175 kg $P \cdot ha^{-1} \cdot year^{-1}$ for run-off to surface water. Since the farm is located in the Po Valley and field slope is negligible, due to a lack of more detailed data about fraction of eroded soil, phosphate emissions through erosion to surface waters were considered negligible. Concerning pesticides, emissions in soil, water and air were assessed following Margni et al. (2002).

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[2] Irrigation efficiency (η_{irr}) is the ration between the volume of water effectively used by the crop and the applied volume per intervention. With low η_{irr}, high water volumes must be applied, but also high amounts percolate and/or evaporate. With high η_{irr} the applied water volumes are close to those effectively used by the crop, therefore, low percolation/evaporation occurs.

[3] The intervention threshold is a precautionary value adopted to define the instant at which irrigating. The threshold value depends on the adopted irrigation method and on water availability meaning the temporal distance between an irrigation turn and the next possible one. When an irrigation turn must be kept into account, a more secure threshold (higher value) is adopted to avoid the risk of an extreme and prolonged water stress that could cause the death of the crop. To calculate when to realise irrigation, this threshold must be included in order to avoid underestimation of irrigation interventions. Conversely, when well water or frequent irrigation turns (1–2 days for Po Valley system) are available and irrigation can be performed almost anytime, the irrigation threshold can be less strict and less precautionary. Commonly, this occurs with sprinkler and especially with drip irrigation.

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Table 3
Life Cycle Inventory for the operations included in the maize grain production, considering also the alternative techniques.

Operation	Repetition	Machinery	Time (h/ha)	Fuel consumption (kg/ha)	Notes	Solution [b]
Organic fertilisers spreading	1	Slurry tanker (20 m^3) 4WD 130 kW tractor	2.60	44.5	With pig slurry; with digestate in S4	BS, S1, S3, S4
	1	Slurry tanker (20 m^3) with 5 anchors 4WD 180 kW tractor	3.20	55.6		S2
Mineral fertilisers spreading	1	Centrifugal spreader 4WD 62 kW tractor	0.50	3.0	Mineral fertiliser (urea and superphosphate)	S5
Ploughing	1	3 ploughshares mouldboard plough 4WD 90 kW tractor	1.66	24.9	35-cm depth	All
Secondary tillage	2	Rotary harrow 4WD 90 kW tractor	1.00	20.2		All
Sowing	1	Pneumatic precision seeder 4WD 62 kW tractor	0.50	8.4	19 kg/ha	All
Chemical weed control	2	Sprayer 4WD 62 kW tractor	0.28	3.3	With 4 kg/ha lumax and 1 + 1 kg/ha dual	All
Top mineral fertilisation	1	Centrifugal spreader 4WD 62 kW tractor	0.50	3.0	Mineral fertiliser (urea)	All
Mechanical weed control	1	Mechanical hoeing 4WD 62 kW tractor	0.83	4.2		All
Irrigation	2–3[a]	Pump 4WD 62 kW tractor	1.10	12.6		All
Harvesting	1	Combine harvester 300 kW	2.00	42.0		All
Grain transport	1	Two trailers, 4WD 90 kW tractor, 4WD 130 kW tractor	2.00	15.1		All
Grain drying	1	Grain dryer	–	191[c]	for 16 t of maize grain with 23% of moisture content	All
Straw management	1	Straw chopper 4WD 90 kW tractor	1.00	18.5		All
Straw collection	1	Straw baler 4WD 90 kW tractor				S3

Notes:
[a] Number of interventions variable according to the year (2011–2015).
[b] Name of the studied solutions in which the single operation is carried out; mainly, except for fertilisers spreading and straw collection, the same operations are completed in all scenarios.
[c] dm^3 of LPG.

With regard to the maximum allowable concentration of the pollutants (Cmax; mg/dm^3), the Italian law on pollutants emissions to water was adopted (D.lgs 152/06 att. 5, 2006), from which the limits were 50.0 mg/dm^3 for NO_3–N and 2.0 mg/dm^3 for PO_4^-. Pollutants natural concentration (Cnat; mg/dm^3) was assumed equal to zero, in accordance with Aldaya and Hoekstra (2010).

Table 4 reports the number of irrigation interventions according to the irrigation method adopted, as well as the Relative Irrigation Supply (RIS) indicator (García Morillo et al., 2015) for water application efficiency.

The number of interventions per growing season has been calculated according to the irrigation water volume and to the irrigation turn schedule. The number of interventions is low with SI, because high water volumes are distributed per intervention. Instead, the interventions increase with PI and, even more, with DI.

For what concerns RIS indicator:

(i) values > 1.2 mean the system is inefficient, as a higher water volume than requested by the crop is distributed. This occurs in SI, when, because of the low technical efficiency (50%), an excessive water volume is used per intervention and a relevant part of this percolates;
(ii) values = 1.0–1.2 are desirable for the optimal water volume application. This occurs with PI, in 2011 with SI and 2012 with DI, since climate conditions and irrigation turn schedule were met;
(iii) values < 1.0 are obtained when the crop grows with a water deficit; this occurs with DI (not in 2012) and in 2013 and 2015 with PI. This stress is not harmful on yield and in DI it is due to irrigation lack during crop ripening and to the low water volumes formerly applied.

4. Results

4.1. Pollution Water Indicator

Table 5 reports the results for the 4 sub-indicators used to build the indicator PWI and for PWI itself while Fig. 3 shows the PWI for the studied scenarios. The PWI is averaged on the 5 years period 2011–2015 to present the results for the analysed period per each scenario.

With regard to WFgrey, phosphate is the substance for which the biggest dilution volume is needed in all scenarios except for S5, where nitrate is the substance that involves a higher dilution volume. In particular, phosphorous is applied in great amounts with organic fertilisers and the maximum concentration of phosphate (2.0 mg/dm^3) allowed in freshwater has a more restrained normative limit than nitrate (50.0 mg/dm^3). Therefore, even if phosphate leaching is quantitatively restrained, the WFgrey can

Table 4
Number of irrigation interventions and RIS indicator per year and irrigation technology adopted.

Years	Surface irrigation (SI)		Sprinkler irrigation (PI)		Drip irrigation (DI)	
	N. interventions	RIS	N. interventions	RIS	N. interventions	RIS
2011	2	1.14	5	1.01	20	0.79
2012	3	1.90	5	1.11	24	1.02
2013	2	1.41	4	0.92	22	0.92
2014	2	2.15	3	1.15	12	0.81
2015	3	1.70	4	0.96	24	0.96

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result higher.

BS, S1 and S2 show the same value for WFgrey (214 m^3/t of maize grain) because the same mass of phosphate is leached (5.028 kg $PO_4^{-}\cdot ha^{-1}\cdot y^{-1}$). S3 shows the highest WFgrey (278 m^3/t of maize grain) due to the higher mass of fertilisers applied to face the additional nutrients removal caused by straw collection; consequently, phosphorous leaching is higher. In S5, nitrate and phosphate leaching are the lowest because the efficiency of mineral fertilisers is higher than the efficiency of organic ones. Therefore, WFgrey is lower (91 m^3/t; −57% respect to BS).

For what concerns freshwater pollution and, in particular, the evaluated impact categories, S5 has the worst environmental behaviour on FE and FEx. For FE the reason is, mainly, the superphosphate production. For FEx, instead, the results are all close to each other because the applied pesticides are the same. The only exception is S3 (−18% respect to other scenarios) because the economic allocation between grain and straw is performed (involving that 18% of the environmental impact is allocated to straw). As regard to ME, S3 is the best option (−75% respect to BS), while S2 is the worst (+33% respect to BS) because of the great nitrate leaching.

Concerning the PWI, the best result is obtained in S3 and S4 (−60% and −54% respect to BS), where ME mainly affects this result. In more details, in S3 straw is harvested after maize grain and, although S3 has the highest WFgrey (due to phosphorous loss), the nutrients present in straw are removed from field and, consequently, do not leach. Concerning S4, leaching is low because of the higher mineral fertiliser equivalency (MFE[4]) of digestate respect to pig slurry.

For BS (slurry incorporation after >72 h), the PWI is better than for S1 and S2 (solutions in which ammonia volatilisation is reduced thanks to fast soil incorporation or direct injection of pig slurry), due to the lower nitrate leaching (responsible for ME). More in details, the PWI in S1 and S2 shows the worst performance (+26% and +30% respect to BS, respectively), mainly due to an increase of ME. For these scenarios (S1 and S2), the spreading techniques (i.e. fast soil incorporation and direct soil injection) permit to steeply reduce NH_3 emissions to air and determine a higher N availability in the soil that partially leaches at the end of the crop cultivation cycle because the nitrogen availability is higher than the crop removal.

Finally, S5 has high environmental loads for FE, ME and FEx, but, thanks to the low WFgrey, it shows a PWI in between the best and worst spreading techniques (−19% respect to BS).

4.2. Water Footprint

With regard to WFA assessment (m^3/t of maize grain), Fig. 4 reports the results for the alternative irrigation methods (SI – irrigation technology adopted in BS, PI and DI) in the 5-years studied period considering the baseline scenario for the fertilisers spreading (slurry spread and incorporated after >72 h). Regarding the three components reported:

(i) WFgreen differs over every year because is affected by precipitation and evapotranspiration, but is the same on the three irrigation methods;
(ii) WFgrey is always the same because, being calculated in accordance with the WFN and being the cultivation method constant and quite standardised all over the analysed area (and in general in Po Valley), grey water is not affected by the irrigation technology;
(iii) WFAblue differs considerably over the years, both because of the climatic annual variability and because of the selected irrigation method. The yearly variability highlights the importance of adopting WFA instead of WF: blue water is the component of WFA that depends on irrigation and when the irrigation technology has a higher efficiency (SI and DI, in particular) WFAblue is lower (on average, between −10% and −43% for PI and between −33% and −60% for DI respect to SI).

Table 5
Absolute values for WFgrey, for FE, ME and FEx impact categories and for the PWI (Pollution Water Indicator) (average results for the period 2011–2015).

Parameters	Unit	Scenario					
		BS	S1	S2	S3	S4	S5
WFgrey	$m^3H_2O\cdot 10^3$/t	0.214	0.214	0.214	0.278	0.170	0.091
FE	kg Peq/t	0.137	0.137	0.137	0.146	0.083	0.158
ME	kg Neq·10/t	0.483	0.644	0.667	0.122	0.219	0.465
FEx	CTUe·10^4/t	0.773	0.773	0.773	0.635	0.773	0.779
PWI	–	**0.306**	**0.386**	**0.397**	**0.122**	**0.142**	**0.271**

Table 6 reports the comparison between WF (WFN approach) and WFA in the considered 5 years.

In all years, WFA results higher than WF, with differences due to climatic variability among years and to blue water assessment method. Considering BS slurry spreading and irrigation systems, WF results are between −21% and −28% respect to WFA. In both WFA and WF methods, irrigation takes place when precipitation is not sufficient for crop water supply. Therefore, in years such as 2014 where green water is high, blue water is low. However, when assessing WFA, considerable differences arise among the three evaluated irrigation technologies, whereas when assessing WF the differences among the three technologies are only due to the different evapotranspiration that takes place according to the irrigation frequency.

Between the two assessment methods of the blue component (WFN and WFA), the following differences would emerge: (i) WFAblue is twice WFblue in SI; (ii) WFAblue is 27% higher than WFblue in PI; (iii) WFAblue is 9% higher than WFblue in DI. In all cases, the difference in blue water between WFA and WF is the biggest for SI, where the irrigation efficiency is the lowest.

5. Discussion

Several studies have been carried out on the Water Footprint quantification of agricultural productions, many of which follow the traditional WFN approach. With regard to maize cultivation, the WFA result in this study is lower than the global average by Mekonnen and Hoekstra (2011) (1222 m^3/t of maize grain) as well as by Huang et al. (2012) (868 m^3/t of maize grain) for China. However, Italian maize production is heavily affected by irrigation, contrarily to studies carried out in other countries. In fact, also Nana et al. (2014) obtained results close to these (479 m^3/t) with a huge role due to blue water (59% of total WF). Concerning water pollution, Zonderland-Thomassen et al. (2014) evaluated the effect of freshwater eutrophication on water pollution, but the study was limited to assessing one impact category. There is consistency in stating that considering only WFgrey causes a lack of information, especially in the agricultural sector where huge amounts of organic fertilisers are spread.

In this study, the impact categories that are mostly influenced by nitrogen and phosphorous released to water and by pesticides were

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[4] MFE is an indicator that informs about the fertiliser ability to supply nitrogen to crops compared with mineral fertilisers that, differently from organic fertilisers, are characterised by the highest efficiency.

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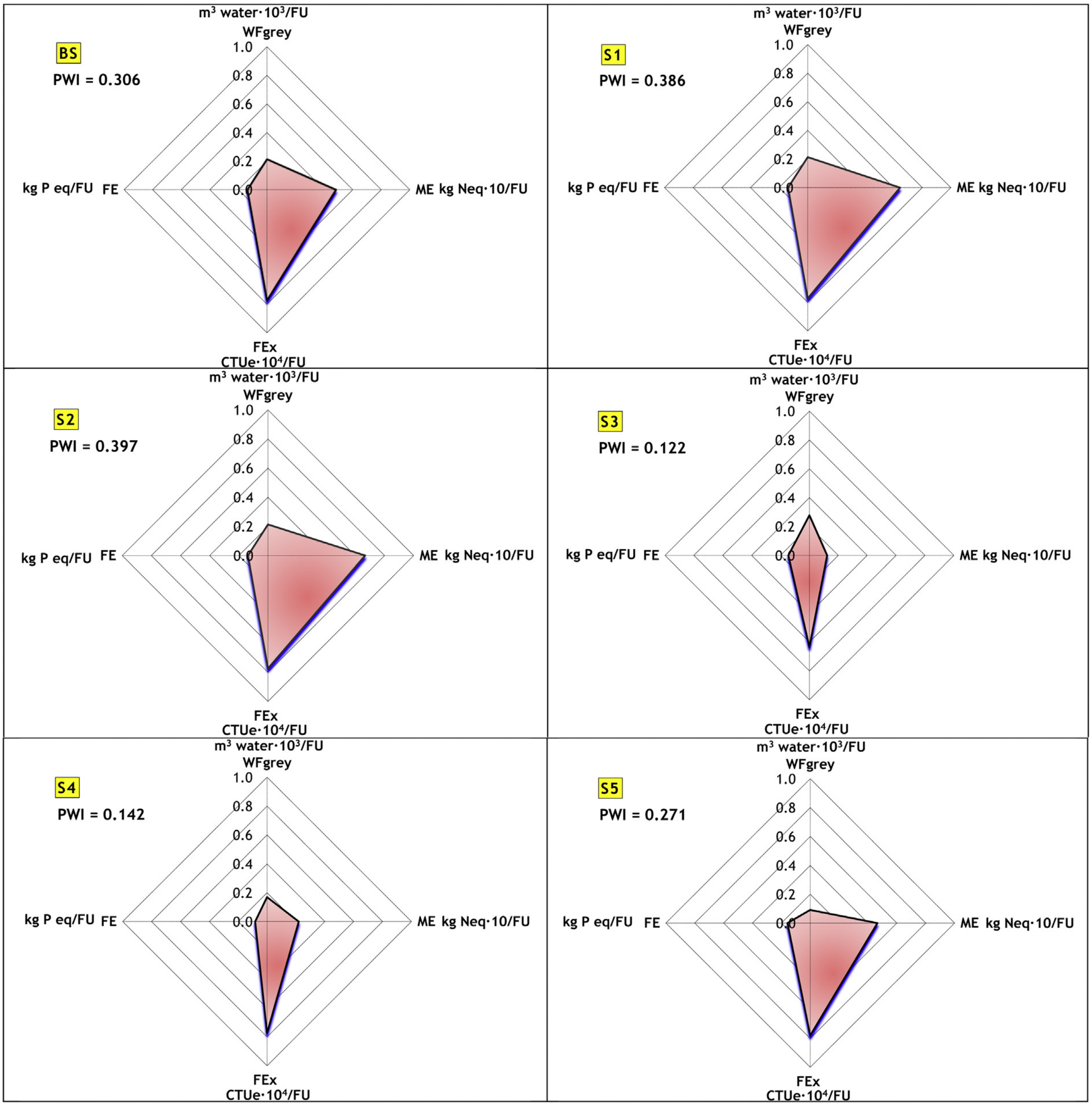

Fig. 3. Pollution Water Indicator (PWI) for the analysed scenarios.

taken into account, together with WFgrey, to develop a new indicator: the Pollution Water Indicator (PWI). It permits to overcome the lacks that characterise WF and to identify the environmental features that affect water pollution not focusing only on one main water pollutant (differently from WF). No former study has been realised with a similar indicator, therefore, no comparison can be done. The main limitation of PWI is that impact categories and WFgrey are put together, though measured with a different unit. To facilitate the comparison among them and make the PWI unit-less, it could be useful to normalise the four sub-indicators selecting either a reference system among the compared ones or an external reference system that permits to carry out the normalisation. At present, at all four sub-indicators has been attributed the same weight, but according to the peculiarities of areas of PWI application it could result notable to introduce a weighting system.

For what concerns irrigation, in the WFN, no difference would occur among the alternative irrigation techniques available to farmers, as no irrigation efficiency is taken into account. According to Hess et al. (2016), the water that returns to the system is used and not effectively consumed (i.e. evapotranspirated), which explains the WFN method. However, it can occur that the consumed water volume is not available enough to sustain the crop water demand (García Morillo et al., 2015) due to the irrigation losses of the adopted method. This determines a lack in water absorbable by the crop. Therefore, it is the gross volume needed for irrigation that must be considered in a crop growing system, because the irrigation efficiency can affect the result. From the application of WFA (volume of used water instead of the consumed one), results show that WFAblue ranges between 41% and 59% of the total water requirement for maize grain production, underling the dependence on irrigation water of maize grain cultivation in Po Valley. The differences between WFA and WFN show consistent yearly variations in a range between 21% and 28%, mainly depending on annual precipitation, crop

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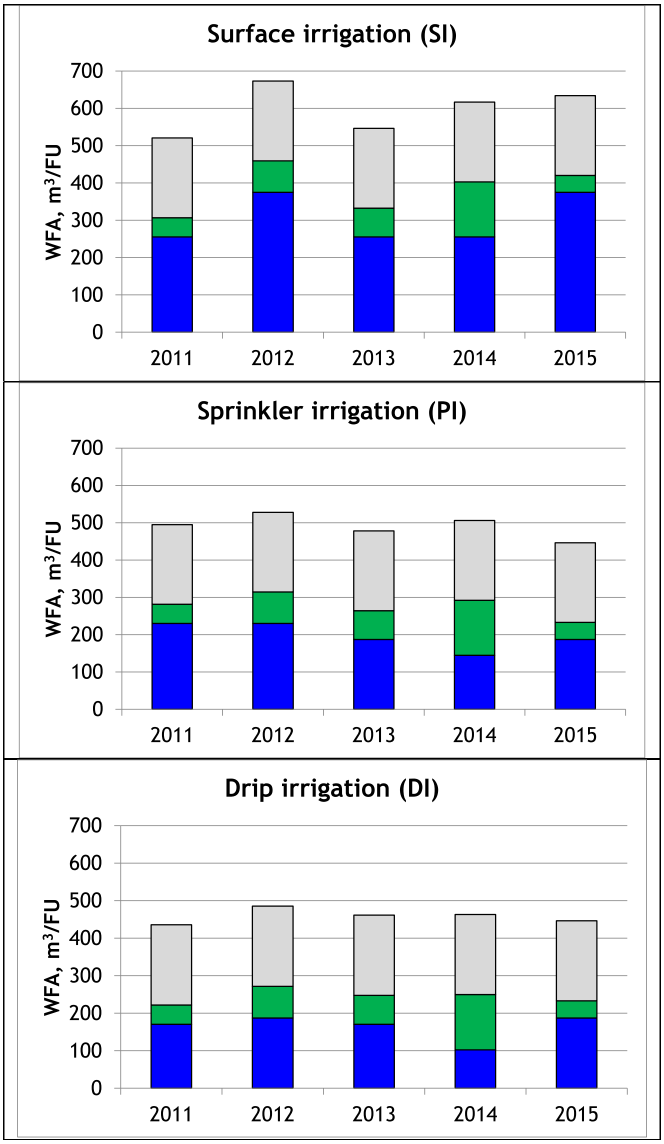

Fig. 4. Water Footprint Applied (blue, green and grey water components) for maize grain production per year considering BS for fertilisers spreading. On the top: surface irrigation (SI). In the middle: sprinkler irrigation (PI). On the bottom: drip irrigation (DI).

evapotranspiration and yield. Surface irrigation is the least efficient technology, but is still frequently adopted in areas with high water availability. If farmers were incentivised to use more efficient technologies as well as inputs more efficiently (Lu et al., 2016), also the WF would reduce.

Table 6
Comparison between total WFA (m^3/t) and WF (m^3/t) (sum of green, blue and grey water) and the relative contribution of blue water in both methods with reference to BS.

Year	WFA m^3/t	WF m^3/t	Blue water			
			WFAblue m^3/t	%	WFblue m^3/t	%
2011	520.82	401.57	255.5	49.1%	136.29	33.9%
2012	673.00	494.12	374.79	55.7%	195.91	39.6%
2013	546.44	427.19	255.54	46.8%	155.03	36.3%
2014	616.67	489.07	255.54	41.4%	127.94	26.1%
2015	634.07	455.22	374.79	59.1%	170.36	37.4%
Average	598.20	453.43	303.24	50.4%	157.10	34.6%

6. Conclusions

The WF is a helpful indicator for the volumetric water requirements of agricultural productions. However, it includes assumptions and simplifications from which critics have arisen (Boulay et al., 2013; Zonderland-Thomassen et al., 2014). To address the main weaknesses of WF a new assessing framework was developed to join both water pollution and consumption. A new indicator of freshwater pollution, called Pollution Water Indicator (PWI), was developed to consider the main water pollutants related to agricultural systems; additionally, the efficiency of the irrigation technique was considered. In fact, respect to the Water Footprint, the new framework does not consider only the pollutant that requires the highest water volume to be diluted, but all the pollutants related to N and P emissions as well as to pesticides application. Besides, all these aspects are synthesised in a single numeric value.

In this study, the WF of maize grain production in Northern Italy was quantified. As concerns grey water, the WFN method was applied and the PWI was developed to evaluate the intensity of water pollution caused by all main polluting substances released to water. Considering PWI, it emerges that WF is not always the best indicator to describe the degradation of water quality, although in volumetric terms. In fact, only considering freshwater environmental impacts and all main substances that have a role on water pollution it is possible to quantify water pollution and identify the related hotspots. Moreover, as regard to blue water, the irrigation efficiency of the adopted irrigation technique must be considered. In fact, the gross volume is the effective amount of water applied on field and its availability affects the crop water supply, crop yield and WF.

The outcomes of the present study can be helpful for policy makers and stakeholders to develop policies, incentives and rules that drive to more efficient and sustainable agricultural systems by allowing understanding the water pollution intensity with a single indicator. In particular, this study is specifically helpful for studying freshwater indicators, improving the use of freshwater resource and of the more efficient technologies for mitigating the environmental impact both due to the demand of high irrigation water and to the detrimental effect on water quality.

Future researches should consider:

(i) the application of PWI to a broad number of crops taking into account also their rotation over the years and, in particular, considering crop rotation with pulse crops or catch crops that can deeply reduce the emissions of N compounds and require low amount of pesticide (Crews and Peoples, 2004; Lemke et al., 2007);
(ii) the assessment of PWI for specific river basins;
(iii) the PWI results variability related to the selection of different methods for assessing the emissions related to fertilisers and pesticides;
(iv) different weights for the 4 sub-indicators that compose the PWI. With this regard, the weights could be varied considering site-specific characteristics. For example, the importance of ME could be enhanced in NVZ (Nitrate Vulnerable Zones constituted in accordance to the European Nitrates Directive) as well as the one of FEx in aquatic ecosystems with high natural value (e.g., river or lake parks).

Authors contributions

All the authors conceived the study and revised the manuscript. DL collected and processed the data, DL and JB wrote the paper.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.12.067.

References

Aivazidou, E., Tsolakis, N., Iakovou, E., Vlachos, D., 2016. The emerging role of water footprint in supply chain management: a critical literature synthesis and a hierarchical decision-making framework. J. Clean. Prod. 37, 1018–1035.

Aldaya, M.M., Hoekstra, A.Y., 2010. The water needed for Italians to eat pasta and pizza. Agric. SyAgric.st. 103 (6), 351–360.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop Evapotranspiration: Guidelines for Computing Crop Water Requirements, FAO Drainage and Irrigation Paper 56. Food and Agriculture Organization, Rome.

Althaus, H.J., Chudacoff, M., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. Life Cycle Inventories of Chemicals. Ecoinvent Report No. 8, v2.0 EMPA. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

ARPA Regional Institution for Protection of the Environment, http://ita.arpalombardia.it/ita/servizi/servizi2.asp. (Accessed 05 June 2016).

Bacenetti, J., Bava, L., Zucali, M., Lovarelli, D., Sandrucci, A., Tamburini, A., Fiala, M., 2016a. Anaerobic digestion and milking frequency as mitigation strategies of the environmental burden in the milk production system. Sci. Total Environ. 539, 450–459.

Bacenetti, J., Lovarelli, D., Fiala, M., 2016b. Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation. Eur. J. Agron. 79, 107–118.

Bacenetti, J., Duca, D., Fusi, A., Negri, M., Fiala, M., 2015a. Mitigation strategies in the agro-food sector: the anaerobic digestion of tomato puree by-products. An Italian case study. Sci. Total Environ. 526, 88–97.

Bacenetti, J., Fusi, A., Negri, M., Fiala, M., 2015b. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Bayart, J.B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A., 2010. A framework for assessing off-stream freshwater use in LCA. Int. J. Life Cycle Assess. 15 (5), 439–453.

Bayart, J.B., Worbe, S., Grimaud, J., Aoustin, E., 2014. The Water Impact Index: a simplified single-indicator approach for water footprinting. Int. J. Life Cycle Assess. 19 (6), 1336–1344.

Blok, V., Long, T.B., Gaziulusoy, A.I., Ciliz, N., Lozano, R., Huisingh, D., Boks, C., 2015. From best practices to bridges for a more sustainable future: advances and challenges in the transition to global sustainable production and consumption: introduction to the ERSCP stream of the Special volume. J. Clean. Prod. 86 (108), 19–30.

Boulay, A.-M.M., Hoekstra, A.Y., Vionnet, S., 2013. Complementarities of water-focused life cycle assessment and water footprint assessment. Environ. Sci. Technol. 47 (21), 11926–11927.

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 (6), 349–357.

Crews, T.E., Peoples, M.B., 2004. Legume versus fertilizer sources of nitrogen: ecological tradeoffs and human needs. Agriculture. Ecosyst. Environ. 102 (3), 279–297.

D.lgs 152/06 Attached 5, 2006. Environmental Regulations. Part III. Rules on Soil Conservation and Combating Desertification, Protection of Waters from Pollution and Water Management.

ERSAF. Regional Authority for services to agriculture and forests. (2016). http://www.ersaf.lombardia.it/servizi/Menu/dinamica.aspx?idSezione=16909&idArea=23152&idCat=23173&ID=23552 (Accessed 15 June 2016).

Fedele, A., Mazzi, A., Niero, M., Zuliani, F., Scipioni, A., 2014. Can the Life Cycle Assessment methodology be adopted to support a single farm on its environmental impacts forecast evaluation between conventional and organic production? An Italian case study. J. Clean. Prod. 69, 49–59.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Humbert, S., Köllner, T., Loerincik, Y., Margni, M., Nemecek, T., 2007. Implementation of Life Cycle Impact Assessment Methods. Ecoinvent Report No. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Fusi, A., Bacenetti, J., 2014. Confronto tra modelli per la stima delle emissioni da fertilizzanti: il caso del mais da granella. In: Proceedings of 8th National Conference of Rete Italiana LCA, 19-20 June 2014, Florence, pp. 168–175.

García Morillo, J., Rodríguez Díaz, J.A., Camacho, E., Montesinos, P., 2015. Linking water footprint accounting with irrigation management in high value crops. J. Clean. Prod. 87, 594–602.

Giardini, L., 2003. A Come Agricoltura. Patron Editore, Bologna.

González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L., 2012. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. J. Clean. Prod. 43, 71–83.

Herath, I., Green, S., Singh, R., Horne, D., Van Der Zijpp, S., Clothier, B., 2013. Water footprinting of agricultural products: a hydrological assessment for the water footprint of New Zealand's wines. J. Clean. Prod. 41, 232–243.

Hess, T., Chatterton, J., Daccache, A., Williams, A., 2016. The impact of changing food choices on the blue water scarcity footprint and greenhouse gas emissions of the British diet: the example of potato, pasta and rice. J. Clean. Prod. 112, 4558–4568.

Hoekstra, A.Y., Hung, P.Q., 2002. Virtual Water Trade. A quantification of virtual water flows between nations in relation to international trade. Int. Expert Meet. Virtual Water Trade 12 (11), 1–244.

Hoekstra, A.Y., 2010. The water footprint: water in the supply chain. Water 93, 12–13.

Huang, J., Zhang, H., Tong, W., Chen, F., 2012. The impact of local crops consumption on the water resources in Beijing. J. Clean. Prod. 21 (1), 45–50.

Ingrao, C., Bacenetti, J., Bezama, A., Blok, V., Geldermann, J., Goglio, P., Koukios, E.G., Lindner, M., Nemecek, T., Siracusa, V., Zabaniotou, A., Huisingh, D., 2016. Agricultural and forest biomass for food, materials and energy: bio-economy as the cornerstone to cleaner production and more sustainable consumption patterns for accelerating the transition towards equitable, sustainable, post fossil-carbon societies. J. Clean. Prod. 117, 4–6.

Ingrao, C., Matarazzo, A., Tricase, C., Clasadonte, M.T., Huisingh, D., 2015. Life Cycle Assessment for highlighting environmental hotspots in Sicilian peach production systems. J. Clean. Prod. 92, 109–120.

ISO 14044, 2006. Environmental management – life cycle assessment –requirements and guidelines. Int. Organ. Stand.

ISO 14046, 2014. Environmental Management – Water Footprint – Principles, Requirements and Guidelines.

ISTAT, 2016. Available at: http://dati-censimentoagricoltura.istat.it/?lang=it. (Accessed 5 June 2016).

Jefferies, D., Muñoz, I., Hodges, J., King, V.J., Aldaya, M., Ercin, A.E., Mila I Canals, L., Hoekstra, A.Y., 2012. Water footprint and life cycle assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. J. Clean. Prod. 33, 155–166.

Jeswani, H.K., Azapagic, A., 2011. Water footprint: methodologies and a case study for assessing the impacts of water use. J. Clean. Prod. 19 (12), 1288–1299.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J., 2007. Life Cycle Inventories of Bioenergy. Ecoinvent Report No. 17. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Lamastra, L., Suciu, N.A., Novelli, E., Trevisan, M., 2014. A new approach to assessing the water footprint of wine: an Italian case study. Sci. Total Environ. 490, 748–756.

Lemke, R.L., Zhong, Z., Campbell, C.A., Zentner, R., 2007. Can pulse crops play a role in mitigating greenhouse gases from North American agriculture? Agron. J. 99 (6).

Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Lema, J.M., Moreira, M.T., 2014. Life Cycle Assessment of electricity production in Italy form anaerobic co-digestion of pig slurry and energy crops. Renew. Energ 68, 625–635.

Lovarelli, D., Bacenetti, J., Fiala, M., 2016a. Water Footprint of crop productions: a review. Sci. Total Environ. 548–549, 236–251.

Lovarelli, D., Bacenetti, J., Fiala, M., 2016b. Effect of local conditions and machinery characteristics on the environmental impacts of primary soil tillage. J. Clean. Prod. 1–13.

Lu, Y., Zhang, X., Chen, S., Shao, L., Sun, H., 2016. Changes in water use efficiency and water footprint in grain production over the past 35 years: a case study in the North China Plain. J. Clean. Prod. 116, 71–79.

Manzardo, A., Mazzi, A., Loss, A., Butler, M., Williamson, A., Scipioni, A., 2016. Lessons learned from the application of different water footprint approaches to compare different food packaging alternatives. J. Clean. Prod. 112, 4657–4666.

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

Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrol. Earth Syst. S. C. 15, 1577–1600.

Milà I Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Clift, R., 2009. Assessing freshwater use impacts in LCA: Part I - Inventory modelling and characterisation factors for the main impact pathways. Int. J. Life Cycle Assess. 14 (1), 28–42.

Milan Grain Association. (2016). http://www.granariamilano.org/pagina.php?id_sottomenu=8&id_menu=2 (Accessed 5 July 2016)

Nana, E., Corbari, C., Bocchiola, D., 2014. A model for crop yield and water footprint assessment: study of maize in the Po valley. Agric. SyAgric.st. 127, 139–149.

Negri, M., Bacenetti, J., Manfredini, A., Lovarelli, D., Fiala, M., Bocchi, S., 2014. Evaluation of methane production from maize silage by harvest of different plant portions. Biomass Bioenerg 67, 339–346.

Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., Schaller, B., Chervet, A., 2011. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agric. SyAgric.st. 104, 233–245.

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Agricultural Production Systems. Final Report Ecoinvent v2.0 No. 15a. Agroscope FAL Reckenholz and FAT Taenikon. Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, Switzerland.

Niero, M., Ingvordsen, C.H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M.F., Hauschild, M.Z., Jørgensen, R.B., 2015. Eco-efficient production of spring barley in a changed climate: a Life Cycle Assessment including primary data from future climate scenarios. Agric. SyAgric.st. 136, 46–60.

Noya, I., González-García, S., Bacenetti, J., Arroja, L., Moreira, M.T., 2015. Comparative environmental profile of cereal cultivation for feed and food production in

Please cite this article in press as: Lovarelli, D., et al., Beyond the Water Footprint: A new framework proposal to assess freshwater environmental impact and consumption, Journal of Cleaner Production (2016), http://dx.doi.org/10.1016/j.jclepro.2016.12.067

Lombardy region (Italy). J. Clean. Prod. 99, 250–265.

Pellegrini, G., Ingrao, C., Camposeo, S., Tricase, C., Contò, F., Huisingh, D., 2016. Application of water footprint to olive growing systems in the Apulia region: a comparative assessment. J. Clean. Prod. 112, 2407–2418.

Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43 (11), 4098–4104.

Prahsun, V., 2006. Erfassung der PO4-Austrage fur die Okobilanzierung SALCA Phosphor. Agroscope Reckenholz – Tanikon ART, pp. 1–20.

Renzulli, P.A., Bacenetti, J., Benedetto, G., Fusi, A., Ioppolo, G., Niero, M., Proto, M., Salomone, R., Sica, D., Supino, S., 2015. Life cycle assessment in the cereal and derived products sector. In: Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds.), Life Cycle Assessment in the Agri-food Sector. Case Studies, Methodological Issues and Best Practices. Springer International Publishing, Switzerland, pp. 185–249.

Repar, N., Jan, P., Dux, D., Nemecek, T., Doluschitz, R., 2016. Implementing farm-level environmental sustainability in environmental performance indicators: a combined global-local approach. J. Clean. Prod. 1–13.

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob. Environ. Change 20 (1), 113–120.

Scarpare, F.V., Hernandes, T.A.D., Ruiz-Corrêa, S.T., Kolln, O.T., Gava, G.J.D.C., dos Santos, L.N.S., Victoria, R.L., 2015. Sugarcane water footprint under different management practices in Brazil: Tietê/Jacaré watershed assessment. J. Clean. Prod. 1–9.

Spielmann, M., Bauer, C., Dones, R., 2007. Transport Services. Ecoinvent Report No. 14. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. International Reference Life Cycle Data System (ILCD) Handbook – towards More Sustainable Production and Consumption for a Resource-efficient Europe. JRC Reference Report, EUR 24982 EN. European Commission – Joint Research Centre. Publications Office of the European Union, Luxembourg.

Wu, B., Zeng, W., Chen, H., Zhao, Y., 2016. Grey water footprint combined with ecological network analysis for assessing regional water quality metabolism. J. Clean. Prod. 112, 3138–3151.

Yoo, S.-H., Choi, J.-Y., Lee, S.-H., Kim, T., 2013. Estimating water footprint of paddy rice in Korea. Paddy Water Environ. 12 (1), 43–54.

Zonderland-Thomassen, M.A., Lieffering, M., Ledgard, S.F., 2014. Water footprint of beef cattle and sheep produced in New Zealand: water scarcity and eutrophication impacts. J. Clean. Prod. 73, 253–262.

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