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The Influence of Fertiliser and Pesticide Emissions Model on Life Cycle Assessment of Agricultural Products: The Case of Danish and Italian Barley

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Abstract

Barley is an ancient crop and a great source of nutrients. It is the third largest agricultural commodity produced in Denmark and represents a relevant crop in Italy too. Due to the increasing customers awareness of sustainability issues, it has become essential to evaluate the environmental impact and the use of resources in food production and distribution systems. However, especially in agriculture, difficulties are encountered when emissions from fertilisers and pesticides need to be modelled, due to a variety of modelling options and their dependency on the availability of site-specific information. How to address these difficulties might affect the results reliability. Hence, this study aims to evaluate, using the life cycle assessment (LCA) methodology, the influence of different models for estimating emissions from fertilisers and pesticides on the environmental impacts of barley cultivation in Denmark and Italy. Two models for fertilisers and pesticides' emissions have been applied; these differ on the extent of data requirements and complexity of calculation algorithms, which might increase the results accuracy and robustness.

The results show that the modelling options do affect the environmental impacts of barley production, in particular climate change, eutrophication categories, acidification and freshwater eco-toxicity. This study estimates that the variations for such categories range from 15% in the case of climate change to 89% in the case of marine eutrophication. These findings highlight the importance of the emission modelling options as well as the constraints of data requirements, critical aspects when a LCA study on agricultural products is carried out.

Key words: Environmental impact, nitrogen and pesticide' emissions, modelling, cereal crops, agricultural systems

1 Introduction

Barley (*Hordeum vulgare* L.) is an ancient crop, extremely adaptable to climate conditions and genetically diverse (Jones et al., 2011; Muñoz-Amatriaín et al., 2014). It is also a great source of nutrients, especially in terms of proteins composition, high concentration of carbohydrates and fibre, among other beneficial nutrients and macronutrients (Baik & Ullrich, 2008). Worldwide, barley is mainly used for animal feedstock and malt production, and to a lesser extent for seed production and human consumption.

Due to its characteristics, barley is the 12th most important agriculture commodity in the world, and Europe is the largest producer, accounting for 62% of the worldwide production (FAO, 2016a). In 2014, the worldwide barley production was estimated at 146.6 million tonnes, equivalent to € 4,836 million (FAO, 2016a; FAO, 2016b). Moreover, according to Eurostat (2016), in 2015 the barley-cultivated area was 12,434,270 ha within the European Union. In the same year, the main European producer was Spain with 2,600,920 ha followed by France (1.764.990 ha), Germany (1,621,800 ha) and Poland (839.300 ha). Significant barley cultivation takes place also in Denmark (631,000 ha) and Italy (237,900 ha).

Barley can be classified by different parameters, being the most common and simple classifications the end use: feedstock or malt; and by cold temperature requirements: spring or winter varieties (Baik & Ullrich, 2008; Anderson et al., 2013; Sullivan et al., 2013). Winter barley requires low temperature to grow, so it is sown in fall and harvested during spring and summer, while spring barley can be sown in spring, because of its flexibility of temperature specifications (Anderson et al., 2013). At European level, spring barley is more widespread respect to winter barley; in fact, it is cultivated over about 59% of the European agricultural area dedicated to barley (Eurostat, 2016), and it is usually cultivated in the Northern countries (Eurostat, 2016). Winter barley is more common in the Mediterranean area; for example, in Italy all the barley area is dedicated to winter barley while in France it is only 74% of the area. Barley is the third largest agricultural commodity produced in Denmark, which accounted for 3.95 million tonnes in 2013, with a net production value of €135.6 million (FAO, 2016a). Although Italy does not have as largest barley production as Denmark, their production is relatively high with around 846,142 tonnes by 2014 (Eurostat, 2016).

In the last decades, consumer awareness related to sustainability issues has significantly increased, and it is expected that consumers would include ecological and ethical aspects in their purchasing decision-making processes in the near future (Pluimers 2001; Poritosh et al. 2009). For this reason, it is essential to provide them with reliable information on the environmental impact and the use of resources in food production and distribution systems. Among all sectors, agriculture has the strongest interaction with nature (Bannayan et al. 2011a,b; Bannayan & Sanjani 2011), and in regions where intensive agriculture is carried out, the contribution of farming systems to the degradation of the environment has been increasingly investigated (Basset-Mens et al. 2006).

Life Cycle Assessment (LCA) plays a key role in the quantification of the potential environmental impacts associated with agricultural production (Notarnicola et al., 2017). Within the crop sector, LCA has been used to quantify the environmental performances of different agricultural systems, such as kiwifruit (Nikkhah et al. 2016), peanut (Nikkhah et al. 2015), legumes (Haugaard-Nielsen et al. 2016), wheat (Laratte et al. 2014, Noya et al., 2015; Fantin et al., 2017), rice (Fusi et al., 2014; Bacenetti et al., 2016a; Fusi et al. 2017, Khoshnevisan et al, 2014, Hayashi et al. 2016), maize (Boone et al. 2016, Bacenetti & Fusi, 2015; Bacenetti er al., 2016b). Several LCA studies have recently quantified the environmental impacts arising from barley production in the Scandinavian countries, focusing on the effects of climate change on future barley cultivation for malting in Denmark (Dijkman et al., 2017; Niero et al., 2015a,b), on the influence of system boundaries definition (Roer et al., 2012), and regional variation in climate and soil organic carbon (SOC) decay of various crops production including barley in Norway (Korsaeth et al., 2014). Moreover, LCA has proved to be useful in forecasting the environmental impacts between conventional and organic barley production in Italy (Fedele et al., 2014) as well as evaluating the life cycle environmental profile of different fodder crops, including barley (Bartzas et al., 2015; González-García et al., 2016), and substrates for bioethanol production (Lechon et al. 2005) under Spanish conditions.

Historical reasons have driven the interest in the assessment of the environmental performance of barley in Denmark and Italy. The production of barley (and wheat) in Northern Europe caused the expansion of cropping areas into formerly un-cropped areas, such as grazing lands and permanent grasslands as well as removal of small woodlands. During the 80s, farmland trees were reduced by around 87%. Intensification of cropping has resulted in increased use of pesticides, with impacts on both pests and non-target wildlife, and also on human health. Moreover, increasing use of fertilisers has resulted in nutrient loss and contamination of ground and surface water through leaching and run-off (Hendy et al., 1995). More generally, the environmental impacts associated with arable farming in European countries include damage to, and removal of soil, the pollution of water sources, and impacts upon biodiversity (Bunzel et al., 2015; Drasting et al., 2016). The deterioration in arable ecosystems is also reflected in the aesthetic quality of the arable landscape (EC, 1999).

One of the main challenges of assessing the environmental performance of agri-food system is modelling emissions from pesticides and fertilisers at inventory analysis (Goglio et al., 2014; Notarnicola et al. 2017). A recent review of LCA studies on cereals performed by Renzulli et al. (2015) pointed out that emissions from fertilisers application are in most cases quantified by the IPCC method (IPCC, 2006) and to a lesser extent by other methods, considering the influence of climatic conditions and cultivation practices, such as the one proposed by Brentrup, et al. (2000). For the estimation of emissions from pesticides, the typical difficulty encountered by LCA practitioners is to quantify the proportion of pesticides emitted to the different media in the ecosphere (i.e. air, soil, water), since usually only the amount applied to the agricultural field is known (Rosenbaum et al. 2015). Currently a number of inconsistent approaches and assumptions are applied in quantifying life cycle emission inventories of pesticides, and so since 2013 a consensus process has been underway to provide recommendations on a consistent accounting of emissions from pesticide and impact assessment in LCA (DTU-MAN-QSA, 2016).

The extensive application of plant protection products (mainly herbicide and pesticides) in combination with wrong agricultural practices could result in environmental issues such as contamination of natural resources and risks for human health (Capri et al., 2007). To reduce exposure from pesticides, the EU Directive on the Sustainable Use of Pesticides (EU128/2009/EC) has been recently revised, and considerable research efforts have been spent to assess the link between attitudes, adoption of self-protective behaviours and exposure (Remondou et al., 2015); and also to assess environmental and human exposure to chemicals (Ciffroy et al., 2016; Sociu et al, 2016). However, environmental threshold values – that are so key element in risk assessment – are not accounted for in LCA studies and therefore this aspect should be carefully considered by LCA practitioners while addressing the impact of pesticide on human health and ecosystems.

The implications of choosing different fertiliser and pesticide emissions models on the LCA of agricultural products have been discussed for wheat, maize and rice (Bacenetti & Fusi, 2015; Bacenetti et al., 2016a; Bacenetti er al., 2016b; Fantin et al., 2017; Fusi et al., 2014; Fusi et al., 2016). However, there are no studies addressing the influence of the estimation of fertiliser and pesticide emissions models for barley. Therefore, the aim of this study is to investigate the influence on the environmental impacts of the application of different models to estimate the emissions from fertilisers and pesticides for barley, considering two cases: Denmark and Italy. This study aims to provide recommendations for LCA practitioners in the agri-food sector on how to address the lack of site-specific data while striving to provide reliable LCA results.

2 Materials and Methods

The environmental impacts of barley production have been assessed using the LCA methodology and following the ISO 14040-44 standards (ISO, 2006a; b). The goal and scope definition is discussed in section 2.1, meanwhile the life cycle inventory (LCI) is reported in sections 2.2 (system description) and 2.3 (allocation). The life cycle impact assessment (LCIA) and sensitivity analysis are discussed in sections 2.4 and 2.5, respectively.

2.1 Goal and Scope Definition

The aim of this study is to evaluate the influence of two different models for fertiliser and pesticide emissions on the environmental assessment of barley production, using Danish and Italian barley as case studies. Hence, a further goal is to estimate the environmental impacts of barley production as feedstock in Denmark and Italy. The outcomes of the study are aimed at barley producers and LCA practitioners.

The functional unit is defined as 'the production of 1 tonne of DM (dry matter) barley used as feed grain'. The scope of the study is from 'cradle to farm gate' including the agricultural field operations; the production of agricultural inputs (i.e. diesel fuel, fertilisers, pesticides, seeds); transport and drying process as well as the emissions from fertilisers and pesticides application. Figure 1 summarises the system boundaries, including the agricultural stages and the main inputs and outputs.

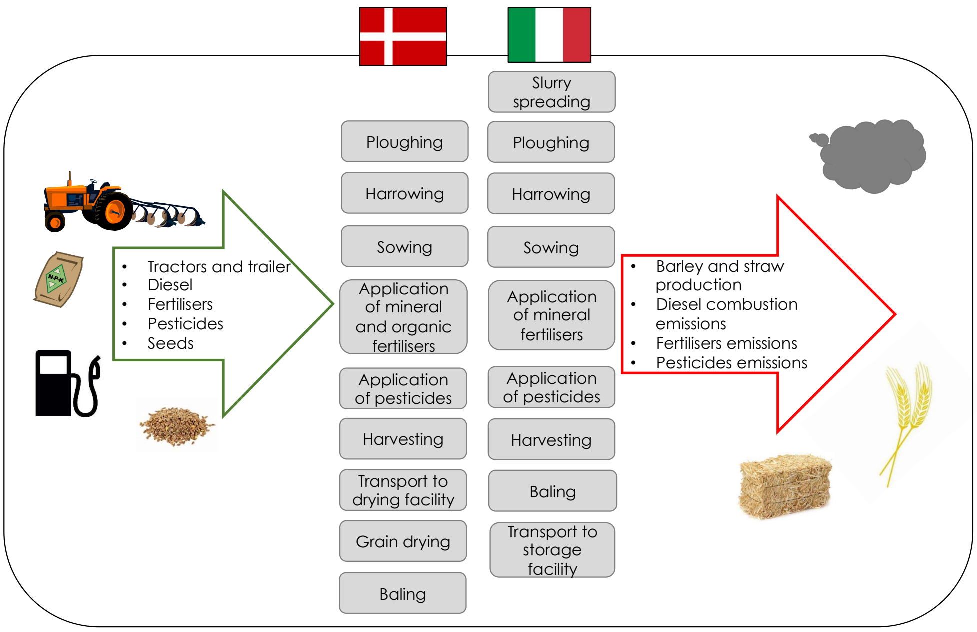


Figure 1 System boundaries of barley life cycle, for both Danish and Italian case studies

2.2 Life Cycle Inventory

In order to facilitate the understanding of the influence of fertiliser and pesticide emissions models, the life cycle stages are grouped as follows:

- Agricultural field operations (including ploughing, harrowing, sowing, chemical weed control, harvesting, straw baling);
- Seeds, fertilisers and pesticides production;
- Grain drying;
- Nitrogen and phosphate (fertilisers)' emissions; and
- Pesticides emissions.

Each sub-system is detailed in the following sections including the field operations and agricultural inputs for each country; moreover, **Table 1** and Table 4 summarise the inventory of the systems.

2.2.1 Agricultural Field Operations

This sub-system includes the field operations carried out in the production of barley, the production, maintenance and disposal of capital goods (tractor and implements) as well as the diesel use and combustions emissions. As seen in Figure 1, the first activity is soil tillage (ploughing and harrowing) for both systems, however in the case of Italy, organic fertilisation is applied previously; the next operation is sowing, carried out in both systems. After this, fertilisation by broadcaster is applied and pest/weed control is carried out using herbicides (1 and 2 interventions in Denmark and in Italy, respectively). Finally, the grains are harvested and transported to the drying/storage facilities, while the straw is bailed and sold. The background information has been sourced from Ecoinvent V.3.1 (Nemecek and Kägi, 2007). Table 1 details the inventories and data sources of both systems.

In terms of transport, it is assumed that the drying facility is within the farm limits, so the distance between the fields and the drying/storage facilities is assumed 1 km, for both barley production systems.

2.2.2 Seeds, Fertilisers and Pesticides Production

As seen in Figure 1, these sub-systems consider critical inputs materials required in the production of barley. The main components are seeds, fertilisers (N, P, K), animal slurry and pesticides (herbicides, fungicides and insecticides). The detailed quantities and specifications for each system are summarised in **Table 1**.

Finally, the transport and packaging of seeds, pesticides and fertilisers are not included in the system boundaries because of lack of data. This is not deemed a limitation as some other studies found their contribution to be insignificant (e.i. Cellura et al., 2012).

2.2.3 Grain Drying

This is the last stage of the production chain, where the grains are transported from the fields to the drying facility to be dried until their moisture content is 14% (commercial moisture). The drying process considers the fuels consumption and the production and maintenance of drying machinery, sourced from Ecoinvent. In the case of Italian barley, due to climatic conditions, drying is not required because the grain already has the commercial moisture after harvested.

Table 1 Life cycle inventory of agricultural practices and inputs for both Danish and Italian case studies [quantities are shown per hectare cultivated (ha); the unit "rep" refers to the number of repetitions]

Inputs	***Unit***	***Denmark***	***Italy***
Yield:			
Grain	t/ha	5.45 [a]	5.05 [b]
Straw	t/ha	4.52 [c]	4.21 [c]
Agricultural field operations:			
Ploughing	rep	1 [d]	1 [b]
Harrowing by rotary harrow	rep	1 [d]	1 [b]
Sowing	rep	1 [d]	1 [b]
Fertilising by broadcaster	rep	1 [d]	1 [b]
Slurry spreading	rep	1 [d]	1 [b]
Pest control application by field sprayer	rep	1 [d]	2 [b]
Harvesting	rep	1 [d]	1 [b]
Bailing	rep	1	1 [b]
Transport (tractor and trailer)	rep	1 [d]	1 [b]
Grain drying	rep	1 [d]	-
Seeds	kg/ha	116 [d]	190 [b]
Fertilisers:			
Calcium Ammonium Nitrate (CAN)	kg N/ha	34 [d]	-
Ammonium Nitrate	kg N/ha	-	70 [b]
Pig slurry	kg N/ha	34 [d]	45 [b]
Dairy cattle slurry	kg N/ha	44 [d]	-
Pesticides:			
Tribenuron-methyl	g/ha	7.50 [d]	12.5 [b]
Pyraclostrobin	g/ha	62.5 [d]	-
Tebuconazole	g/ha	62.5 [d]	-
Pirimicarb	g/ha	62.5 [d]	-
Difensulfuron	g/ha	-	12.5 [b]
Bromoxinil	g/ha	-	238 [b]
2,4 D	g/ha	-	238 [b]

[a] Source: Statistics Denmark (2010-2014): 5 years average
[b] Source: Farms surveys and Italian granary association, AIC (2016) and Associazione Granaria di Milano (2016)
[c] Calculated from HI and round baler efficiency
[d] Source: Niero et al. (2015b)

2.2.4 Emissions from Fertilisers and Pesticides

The development and analysis of the life cycle inventory, second stage of the LCA framework (ISO, 2006a; b), is very critical for an LCA study. The accuracy, robustness and reliability of the results depend on the quality and representativeness of the data. Hence, the accountability of not only the material and energy input-outputs flows, but also the emissions are crucial. That is why, in the case of agri-food systems, the high variability and lack of consensus in terms of critical variables as fertilisers and pesticides modelling options need to be studied, and hopefully best practices will be set soon. A specific review (**Error! Reference source not found.**2) considering: a) LCA studies on barley; b) the latest barley and other crops LCA studies published by STOTEN; and c) some of the latest published LCA studies including crops and agri-food systems, showed that IPCC (2006) is the most common model used to estimate emissions from fertilisers. For pesticide emissions models, the most interesting finding is the lack of transparency, since more than half of the studies did not specify whether and how pesticides emissions were considered and/or quantified.

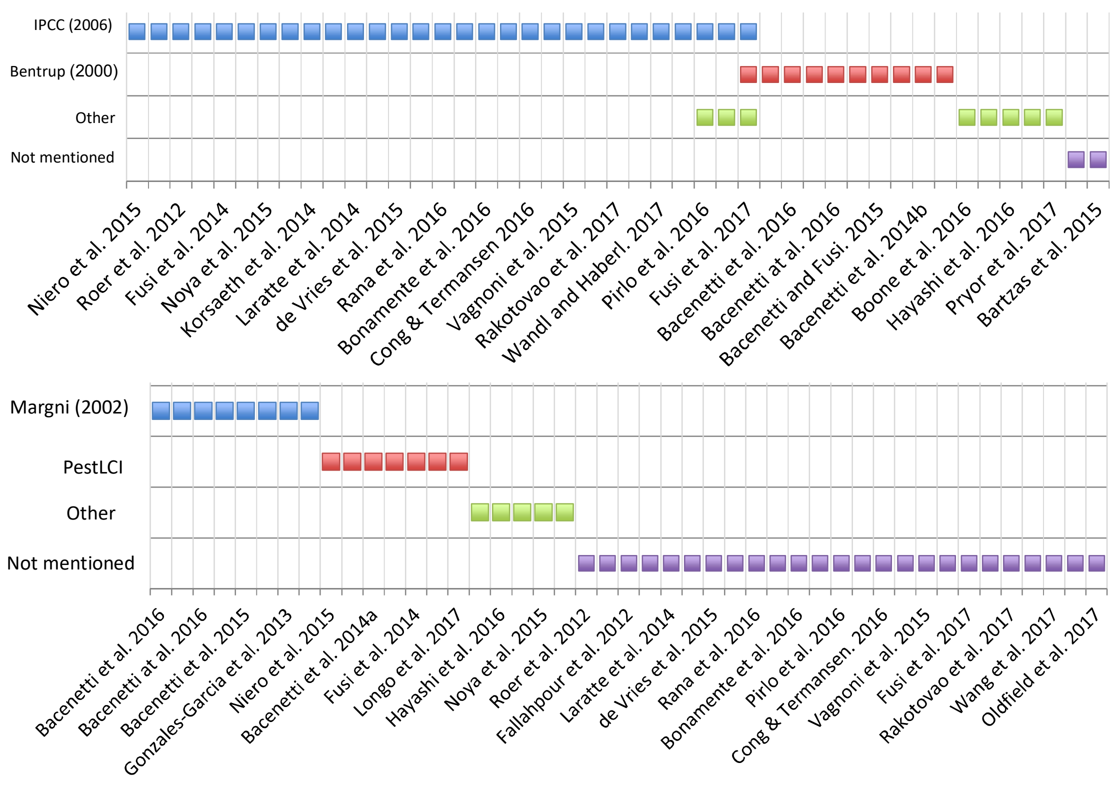


Figure 2 Summary of the rapid evidence assessment of the fertiliser and pesticide emissions models use in Agri-food LCA studies [in the figure, PestLCI refer to both PestLCI 2.0 and PestLCI]

This study considers two scenarios, named baseline (BS) and alternative (AS) scenarios, which examine two models to calculate emissions from fertilisers and pesticides, respectively. The selected models for fertilisers' emissions are IPCC (2006), i.e. the most used fertilisers' emission model in agricultural LCA (Renzulli et al. 2015), and a more detailed model proposed by Brentrup (2000), which is increasingly used by LCA practitioners (see Figure 2).

For pesticide emissions in LCA, the most simplified approaches assume that pesticides are entirely emitted in the soil compartment, as it is done in one of the most used LCA databases, i.e. Ecoinvent (Nemecek and Kägi 2007) or that 85% is release to soil, 10% run-off from the soil into the water, 5% to crops, and 10% to air (Margni et al., 2002; Audsley et al., 2003). This last approach has been used in a large number of agricultural LCA studies (see Figure 2). A different approach is considered by PestLCI, a model for estimating field emissions of pesticides based upon fate and exposure modelling principles as applied in relation to risk assessment of single chemical substances (Birkved and Hauschild, 2006). PestLCI and its latest version, PestLCI2.0, estimate emissions to three general environmental compartments: air, surface water and groundwater. PestLCI2.0 considers that after the primary distribution of pesticides over leaves and soil has taken place, then three secondary fate processes on leaves occur: volatilization, degradation and uptake, as explained in details by Dijkman et al. (2012). The use of PestLCI2.0 has recently increased for LCA of cereals, i.e. barley (Niero et al. 2015a,b, Dijkman et al. 2017), wheat and maize (Bacenetti et al. 2014, Fantin et al. 2017).

From LCA practitioners´ point of view, the main differences between the methods are the level of detail information in terms of on-site data required to apply each of them. The baseline scenario (BS), IPCC (2006) and Margni et al. (2002), does not require detailed on-site data to be applied, just the amount of fertilisers (N, P, K) and pesticides (active ingredient). Meanwhile the alternative scenario (AS), Brentrup et al. (2000) and PestLCI 2.0 (Dijkman et al. 2012), depends on site-specific information (e.i. type of soil, temperatures, wind, etc.), which is not always easy to obtain. However, estimations based on site-specific information are supposed to be more accurate. Table 2 summarises the scenarios and models included in this study while a detailed description of each is explained below.

Table 2 Scenarios and the fertilisers and pesticides emissions models used

Scenarios	***Fertiliser emissions model***	***Pesticide emissions model***
Baseline (BS)	IPCC (2006)	Margni et al. (2002)
Alternative (AS)	Brentrup et al. (2000)	PestLCI 2-0 (Dijkman et al. 2012)

- ***Fertiliser Emissions Models***

The information required to apply the algorithms defined by the IPCC (2006) is the nitrogen content in the organic and mineral fertilisers applied, while the model developed by Brentrup et al. (2000) requires specific on-site information in order to be applied, such as:

1. The temperature, precipitation and the wind speed at the time of distribution of the fertiliser;

2. The time between the fertilisers distribution and its incorporation into the soil;
3. The soil characteristics (e.g. texture, pH, soil organic matter, cation exchange capacity, etc.);
4. Summer and autumn rainfall; and
5. The content of nitrogen in the harvested grain and co-product.

In this model, the calculation of nitrogen emissions is based on the difference between the N supplied and the N absorbed, the characteristics of the soil, the climate conditions, the type of fertilisers used and the application method. The list of input parameters for the Danish and Italian cases is reported in Table 3. The software EFE-So (2015) has been used to calculate the emissions defined by Brentrup's algorithms.

Very often, all or part of the data listed above are not available to LCA practitioners, and it is therefore necessary to use a simplified model, like the one proposed by the IPCC (2006), which only requires to the nitrogen content in the organic and mineral fertilisers applied.

Table 3 List of main input parameters included in the model by Brentrup et al. (2000)

Parameter	***Denmark***	***Italy***
Characteristics of the organic fertiliser	Pig slurry[a]: dry matter content 1.89%, pH 7.5, total N content 2.43 $kg \cdot t^{-1}$, ammonia content 0.75 $kg \cdot t^{-1}$, P content 2.1 $kg \cdot t^{-1}$; high infiltration rate	Pig slurry[a]: dry matter content 1.89%, pH 7.5, total N content 2.43 $kg \cdot t^{-1}$, ammonia content 0.75 $kg \cdot t^{-1}$, P content 2.1 $kg \cdot t^{-1}$; high infiltration rate
Air temperature during spreading	5-10°C	10-15°C
Soil texture	Medium texture	Medium texture
Rainfall after the spreading	5 mm	No rain in the first 3 days
Atmospheric deposition of N	15.6 $kg \cdot ha^{-1}$ during crop cultivation	22.5 $kg \cdot ha^{-1}$ during crop cultivation
N content in grain	1.9% of dry matter [b]	1.9% of dry matter [b]
N content in straw	0.6	0.7 of dry matter [b]
Rainfall in winter season	800 mm	370 mm
NH_3 emission factor for mineral fertilisers	CAN: 1% of total applied mineral N [c]	Ammonium nitrate: 2% of total applied mineral N [c]

[a] Source: Lijò et al., (2014) and Lijò et al., (2015)
[b] Source: Baldoni and Giardini (2000)
[c] Source: Brentrup et al. (2000).

For both scenarios, phosphate emissions into water were calculated following Nemecek and Kägi (2007). Two different phosphorus emissions into water were considered: leaching into the ground water and run-off into surface water.

- ***Pesticide Emissions Models***

In their study, Margni et al. (2002) proposed the share percentages by which the active ingredient of pesticides should be multiplied in order to obtain their distribution into the plant, soil, water and air compartments, i.e. 5%, 76.5%, 8.5% and 10%, respectively. While this approach is quite easy to implement, the application of PestLCI 2.0 requires much more data. Such information relates to:

1. Pesticide type;
2. Crop type and stage of development of the crop (e.g. leaf development, tilling, booting/senescence);
3. Soil type and climate;
4. Month of application; and
5. Application method.

The list of input parameters and output from the application of these methods on the baseline and alternative scenarios are displayed in Table 3 and Table 4, respectively.

Table 3 List of main input parameters used in the PestLCI 2.0 model

		Denmark				***Ital***
Pesticide	***Crop development stage***	***Soil type***	***Climate***		***Pesticide***	***Crop development stage***
Tribenuron-methyl	Cereals I - leaf development	JB-6 (Sandy loam, 1324 Vinderslev DK)	02 - Temperate maritime I: Tranebjerg (DK)		Tribenuron-methyl	Cereals I - leaf development
Picoxystrobin	Cereals II - tillering	JB-6 (Sandy loam, 1324 Vinderslev DK)	02 - Temperate maritime I: Tranebjerg (DK)		Thifensulfuron-methyl (Difensulfuron)	Cereals I - leaf development
Tebuconazole	Cereals IV - booting/senes cence	JB-6 (Sandy loam, 1324 Vinderslev DK)	02 - Temperate maritime I: Tranebjerg (DK)		2, 4 D	Cereals II - tillering
Pirimicarb	Cereals IV - booting/senes cence	JB-6 (Sandy loam, 1324 Vinderslev DK)	02 - Temperate maritime I: Tranebjerg (DK)		Bromoxynil	Cereals II - tillering

Table 4 Fertiliser and pesticide emissions models used in the baseline (BS) and alternative (AS) scenario

		Denmark								
		BS[a] [kg/ha]			***AS[b] [kg/ha]***			***BS[a] [kg/ha]***		
		Air	Soil	Water	Air	Soil	Water	Air	Soil	W
Fertilisers' emissions	NH_3	23.19	-	-	7.02	-	-	31.41	-	
	N_2O	2.64	-	-	2.03	-	-	3.12	-	
	NO_3	-	-	161.21	0	-	-	-	-	
	PO_4	-	-	0.299 (gw) 1.276 (sw)	-	-	0.299 (gw) 1.276 (sw)		-	

										2.568 (sw)			
Pesticides' emissions	Tribenuron-methyl	7.50E-4	6.38E-3	7.50E-4 (gw)	6.93E-5	-	1.86E-4 (gw)	1.25E-3	9.56E-3	1.06E-3	2.13 E-4	-	17.5E-4 (gw) 7..25E-5 (sw)
	Pyraclostrobin	6.25E-4	-	6.25E-4 (gw)	2.39E-4	-	7.97E-6 (gw) 1.46E-7 (sw)	-	-	-	-	-	-
	Tebuconazole	6.25E-4	5.31E-2	6.25E-4 (gw)	1.79E-5	-	2.20E-9 (gw) 1.23E-6 (sw)	-	-	-	-	-	-
	Pirimicarb	6.25E-4	5.31E-2	6.25E-4	5.77E-5	-	1.89E-8 (gw) 1.50E-5 (sw)	-	-	-	-	-	-
	Thifensulfuron-methyl (Difensulfuron)	-	-	-	-	-	-	1.25 E-3	9.56 E-3	1.06E-3	2.13E-4	-	17.5E-4 (gw) 8.0E-5 (sw)
	2, 4 D	-	-	-	-	-	-	28.0 E-3	214 E-3	23.8E-3	1.29E-3	-	1.34 (gw) 39.2E-4 (sw)
	Bromoxynil	-	-	-	-	-	-	28.0E-3	214 E-3	23.8E-3	3.36	-	0.11 (gw) 2.16E-3 (sw)

[a] Fertilisers sourced from IPCC (2006) and pesticides from Margni et al. (2002)
[b] Fertilisers sourced from Brentrup et al. (2000) and pesticides from PestLCI 2.0 (Dijkman et al. 2012)
gw = groundwater; sw = surface water

2.3 Allocation

Besides barley grain, the cultivation phase originates as a co-product, straw. The latter is usually chopped and incorporated into the soil to improve quality, but it could also be used as animal bedding or for energy generation. Consistently with other LCA studies facing multi-functionality in agriculture (e.g. Andersson, 2000; Fallahpour et al., 2012; Fusi et al., 2014), an economic-based allocation method has been implemented. Table 5 reports the market prices and the allocation factors considered for each co-product.

The amount of straw produced was calculated considering a Harvest index (HI) of 0.48, according to Baldoni and Giardini (2000). Considering that the basal portion of barley culms cannot be collected, only 85% of the straw produced has been considered collectable. Besides this, 90% collection efficiency has been taken into account for round-baler. Therefore, only 76.5% of the straw is collected. Table 5 summarises the allocation factors use in the Danish and Italian cases.

Table 5 Prices and economic allocation factors for straw for Danish and Italian case studies

Parameter	***Denmark***	***Italy***
Average barley grain price [€/t]	179 [a]	180 [c]
Average straw price [€/t]	24 [b]	70 [b]
Allocation factor barley grain [%]	90	76
Allocation factor barley straw [%]	10	24

[a] 1.335 DKK/kg 5 years average (2010-2014) (Statistics Denmark), conversion rate 1€=7.44 DKK (XE, 2016).
[b] 0.179 DKK/kg 5 years average (2010-2014) (Statistics Denmark)
[c] Source: AIC (2016)

2.4 *Life Cycle Impact Assessment*

The impact assessment has been estimated according to ILCD 2011 methodology v1.05 (Hauschild et al. 2013), using SimaPro software (PRé Consultants, 2015). Emissions from fertilisers' application are particularly relevant for impact categories such as acidification (NOx, NH_3) and eutrophication (NOx, NH_3, NO_3, PO_4) (Renzulli et al. 2015), meanwhile emissions from pesticides application influence the toxicity-related impact categories. Moreover, field emissions from fertilisation (N_2O) affect the greenhouse gas (GHG) emissions (Peter et al., 2016) and therefore climate change impact category. However, since this study also seeks to assess the environmental impacts of Danish and Italian barley as feedstock, a full set of impacts are assessed: climate change (CC, kg CO_2 eq.), ozone depletion (OD, kg CFC-11 eq.), human toxicity, considering both cancer effects (HT-c, CTUh) and non-cancer effects (HT-nc, 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 (FET, CTUe), mineral, fossil & renewable resource depletion (MFRD, kg Sb eq.).

2.5 *Sensitivity analysis and validation*

To test the robustness of the LCA results two sensitivity analyses (SCs) were performed to address the following issues:

- *Scenario analysis 1 (SC1) - Influence of fertiliser type.* This scenario uses only mineral fertilisers (100%) to assess how the different models are affected, and therefore how the environmental impacts would change if organics fertilisers were not used (Bacenetti et al., 2016b; Fusi et al., 2016); and
- *Scenario analysis 2 (SC2) - Inclusion of NOx emissions.* This scenario aims to explore how NOx would affect the environmental impacts of the different cases. NOx emissions are modelled according to Hamelin et al. (2012).

Furthermore, the environmental impacts estimated in this study, for both Danish and Italian barley, are validated in section 3.3 through a comparison with available secondary data of average barley producers in several European countries.

3 Results and Discussion

The results of the environmental impacts assessment of the Danish and Italian barley production are discussed in Section 3.1, meanwhile the results of the sensitivity analyses are reported in Section 3.2, where first the influence of the type of fertilisers is assessed (SC1), and then the NOx emissions are evaluated (SC2). Finally, a validation and comparison of the results is carried out (section 3.3.).

3.1 LCA of Danish and Italian Barley Production Using Baseline and Alternative Modelling Options

This section analyses the influence of the fertilisers and pesticides' emissions models on the environmental impacts of two barley production systems: Danish and Italian. First, the results for the Danish system will be discussed, followed by the Italian system, both comparing the baseline and alternative scenarios.

3.1.1 Danish Barley Production

As seen in Figure 3, only five impacts, CC, TA, TE, ME and FET, are affected by the two modelling options; CC is the slightest affected, with variations of 7%, mainly due to reductions in N_2O emissions from fertilisers (Nitrogen emissions) when the alternative model (AS) is used. In the case of TA and TE, both decrease by 58% and 61%, respectively. These impacts are mainly driven by the ammonia emissions (NH_3) from fertilisers (nitrogen emissions), which are reduced by 66% when AS is emplaced. The fourth impact is ME, here AS shows higher values (89%), due to reductions of the estimated nitrogen emissions from fertilisers, in particular nitrate (NO_3) and ammonia (NH_3), which decrease by 96% and 66%, respectively. Finally, FET experiences reductions of 24%, due to the pesticides' emissions decrease by 85% with AS.

Figure 3 also shows the contribution of the different life cycle stages to the Danish barley production. Fertilisers' production and field operations stages are the main contributors across all impact categories, in accordance with previous LCA studies on Danish barley (Niero et al. 2015b, Dijkman et al. 2017). The former plays a main role in

HTnoc, POF and MFRD, being responsible for more than half of the impacts (65% on average). Agricultural field operations are also important contributor of HTc and ODP (30%). Moreover, the fertilisers' production stage has a great impact on HTc (56%), OD (56%), CC (45%) and POF (43%), and an important contribution in MFRD (32%) and FET (38%).

Although the field operations stage is not affected by the modelling options, this stage is one of the major contributors to all the environmental impacts, in particular in those affected by diesel use and its combustion; for instance, the field operations are responsible for 80% of HTnc, 50% of POF and around 66% of MFRD. Finally, this stage is important in OD and HTc (30% each) as well as in CC (16%) and FET (~20%). In the case of Danish barley, the field operations include ten activities, but only five of them account for ~95% of the impacts caused by this stage. Harrowing and ploughing are the greatest contributors with an average contribution of 33% and 24% across all the impacts. Then, fertiliser's application and harvesting are responsible for on average 15% and 17% of the impacts of the stage. Finally, sowing and harrowing add on average 6% of the impacts. The other operations contribute less than 1% each.

As mentioned, fertilisers' emissions also play an important role; specifically, this stage influences CC, TA, TE and ME. Their contribution and variation due to modelling have been already explained. Moreover, emissions from pesticides only influence FE (24%) when the baseline model (BS) is used.

Phosphate emissions only affect FE, being the main contributor with 74%; it is important to remark that phosphate emissions are not affected by the two modelling options.

Overall, seeds production does not show an important contribution to any impact except for FET, where adds ~26% of the impacts. Similar, grain drying slightly influences the results, with contributions fluctuating from 0% to 11%. Finally, pesticides production shows a negligible contribution, with less than ~1% across all the impacts

3.1.2 Italian Barley Production

Figure 4 shows the results of the baseline (BS) and alternative (AS) scenarios of winter barley grown in Italy. The same five impact categories (CC, TA, TE, ME and FET) as in Danish case are affected by the two modelling options. AS shows lower values compared to BS, with reductions ranging from 16% (CC) up to 85% (ME). Nitrogen emissions from fertiliser's application, namely N_2O, NH_3 and NO_3, are lower when the model by Brentrup et al. (2000) is adopted instead of the IPCC one (2006). In turn, the latter determines lower impact in CC, TA (-24.1%), TE (-24.2%%) and ME. In addition, a decrease of 45% is highlighted in the FET category, due to the pesticides emissions drop in AS.

From the hotspots analysis also shown in Figure 4, fertilisers' emissions (both N and P) and field operations are the main contributors. The former plays a critical role in CC, TA, TE, FE and ME, where their contribution is, respectively, 49% (BS) and 39% (AS), 92%

(BS) and 89% (AS), 91% (BS) and 89% (AS), 97% (BS and AS), and 89% (BS) and 27% (AS). The field operations stage is the main contributor to the impacts affected by diesel combustion, such as CC (36% and 43% in BS and AS respectively), OD (86%), HTc (68%), HTnc (92%), POF (92%) and MFRD (92%). The field operations stage includes nine major activities, the most relevant of which are: harrowing, with an average impact of 26%, fertiliser application, accounting for 22% of the impacts and ploughing which contributes on average 16%.

Pesticides' emissions only influence FET (45%) when the baseline model is used. Seeds production only affects FET, where it adds 30% (BS) or 45% (AS). The same goes for fertilisers' production, the contribution of which is maximum 17% in HTc. Grain drying and pesticides production play a negligible role in all the impact categories taken into account.

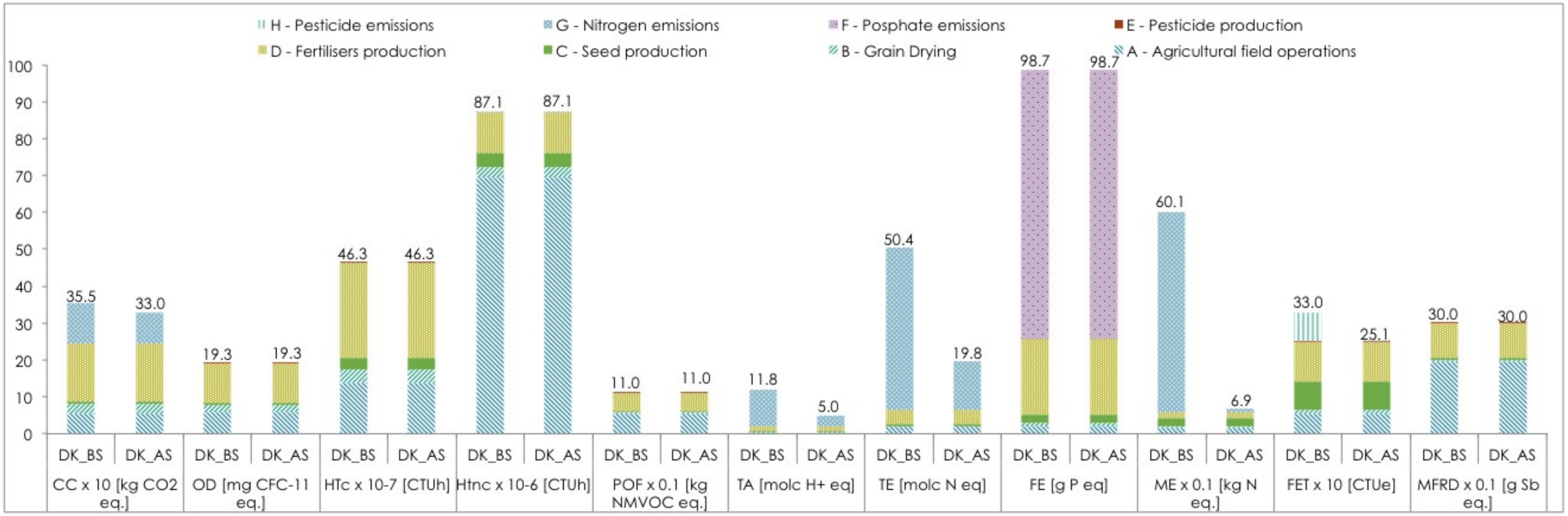


Figure 3 Comparison of the environmental impacts of the baseline (BS) and alternative (AS) scenarios for the Danish (DK) barley production [Results are scaled to fit, factors in x-axis should be multiplied by values of each impact; impacts acronyms can be seen in section 2.4]

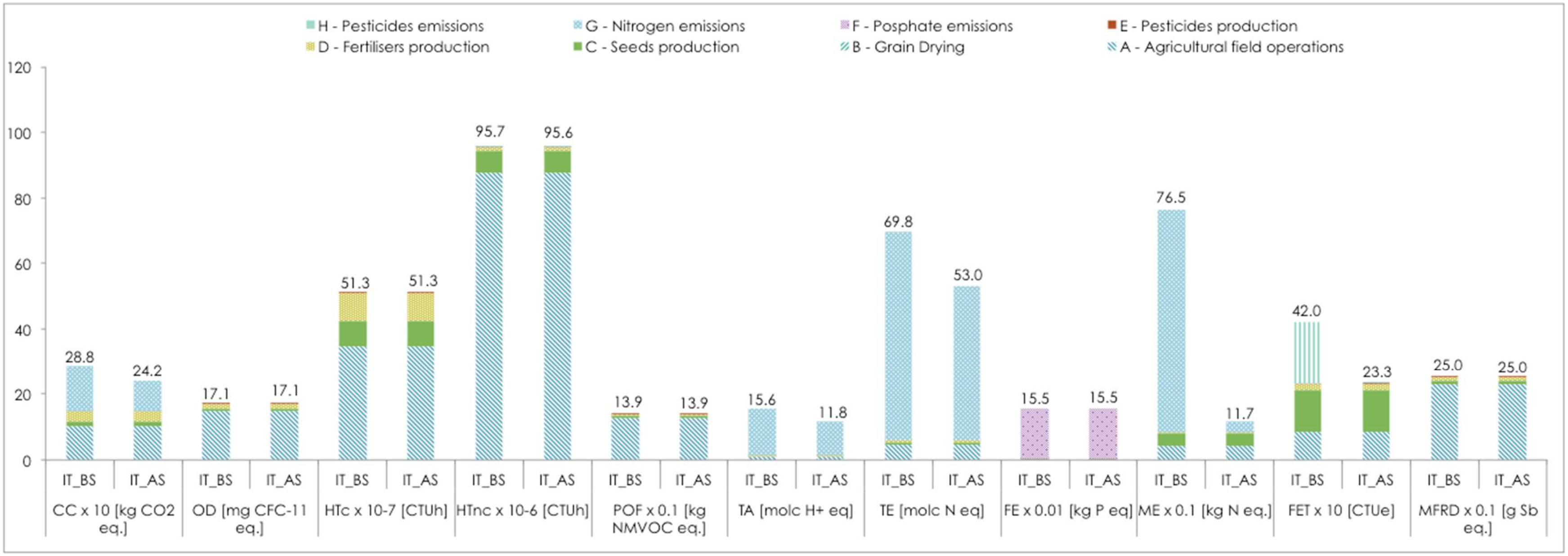


Figure 4 Comparison of the environmental impacts of the baseline (BS) and alternative (AS) scenarios for the Italian (IT) barley production [Results are scaled to fit, factors in x-axis should be multiplied by values of each impact; impacts acronyms can be seen in section 2.4]

3.2 Sensitivity Analyses

This section shows the results estimated from the two sensitivity analyses carried out. First, the influence of type of fertiliser (SC1) is discussed, followed by the influence of accounting NOx emissions (SC2).

3.2.1 SC1: Influence of Type of Fertiliser (100% mineral fertiliser)

- **Danish Barley Production**

As shown Figure 4a, when only mineral fertilisers are used and the system is analysed using BS model, five impacts decreased their score: CC (1.5%), ME (2%), TA (34%), TE (36%) and FE (39%). Small reductions in the estimated nitrous oxide (N_2O) emissions (5%) directly influence the slight variations in CC and ME; alike, the greatest reductions in the estimation of ammonia (NH_3) (41%) and phosphate (52%) are the main responsible for the larger improvements in acidification and eutrophication related impacts (TA, TE, FE and ME).

When AS is emplaced (see Figure 5a), although the same impacts are affected, the changes are different. The increment in nitrous oxide emissions by 6%, directly affects climate change, which increases by 1.5%. Similarly to BS, ammonia (NH_3) and phosphate emissions are much lower when only mineral fertilisers are applied. However, AS estimates much lower emissions (84% and 52%, respectively) than BS, notably decreasing the acidification and eutrophication. Specifically, TA decreases by 50% while TE and FE cut down by 57% and 39%, respectively. Finally, although slightly, ME impact score improves by 11%.

- **Italian Barley Production**

The results of this sensitivity analysis are shown both for BS and AS modelling options in Figure 4b. The adoption of only mineral fertilisers determines an increase of 13% (and 31% in AS), 16%, 46% and 9% (and 17% in the AS) in the categories of CC, OD, HTc and FET, respectively. Such variations are mainly due to the higher energy requirement for producing mineral fertilisers rather than using manure. On the other hand, the application of mineral fertilisers entails a reduction of the environmental impacts in HTnc, TA, TE, FE and ME. The reduction of these impacts ranges from 8% (HTc) to 73% (FE) both in the baseline and the alternative-modelling scenario. The TA and TE categories improve by approximately 55% and 79% in the baseline and the alternative scenario respectively, while ME decreases by 28% and 26%. This is due to a reduction of the N emissions associated with mineral fertilisers application.

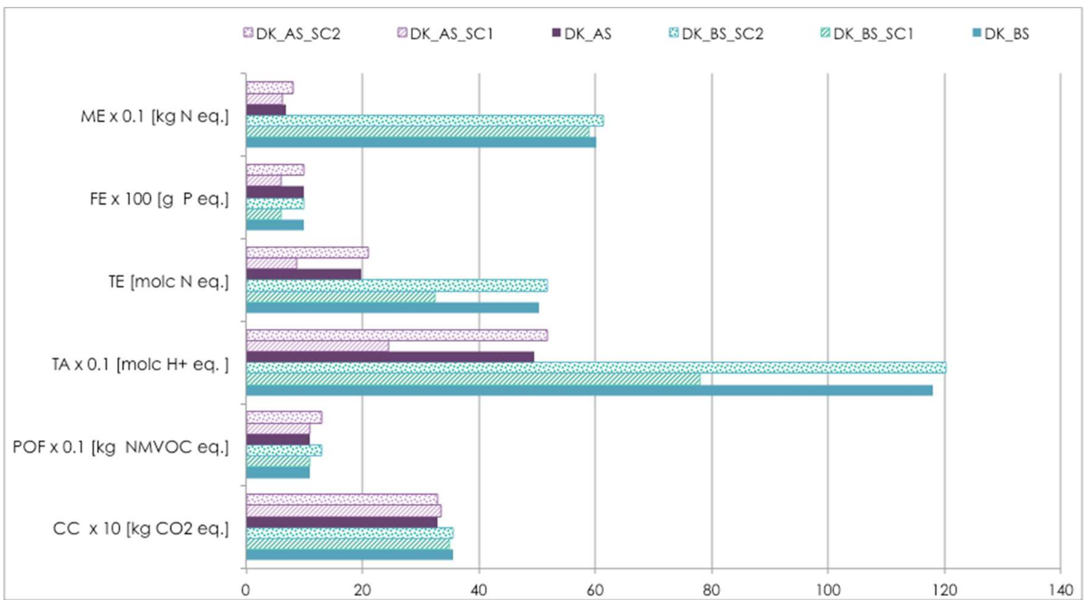
3.2.2 SC2: Inclusion of the NOx Emissions

- **Danish Barley Production**

Figure 4a shows the sensitivity analysis of the inclusion of NOx emission and the influence on the Danish barley production. Similar as the previous analysis, the NOx model only affects four impact categories: POF, TA, TE and ME. In the case of BS, the greatest increment is seen in POF, where the NOx emissions deteriorate this impact by 18%. The other three impacts, TA, TE and ME, are less affected, increasing by less than 3%, as NOx has lower influence on these impacts. Alike, when AS is analysed the complementary NOx emissions affect the same impacts as before, however on different magnitudes; the only exception is POF, where the increments are the same, 18%. In the case of TA and TE, the inclusion of NOx increases the impacts by 4% and 6%, respectively. Finally, the greatest variation is found in ME, where the NOx modelling shows a much larger influence, increasing ME by 17%.

- **Italian Barley Production**

As seen in the Danish case study, the inclusion of NOx emissions only affects four impacts categories (see Figure 5b). POF is affected by 7.1% (BS) and 4.8% (AS), and ME by 0.5% (BS) and 2.2% (AS). The effects on TA and TE are negligible. Overall, the influence of NOx is quite small both in BS and AS.



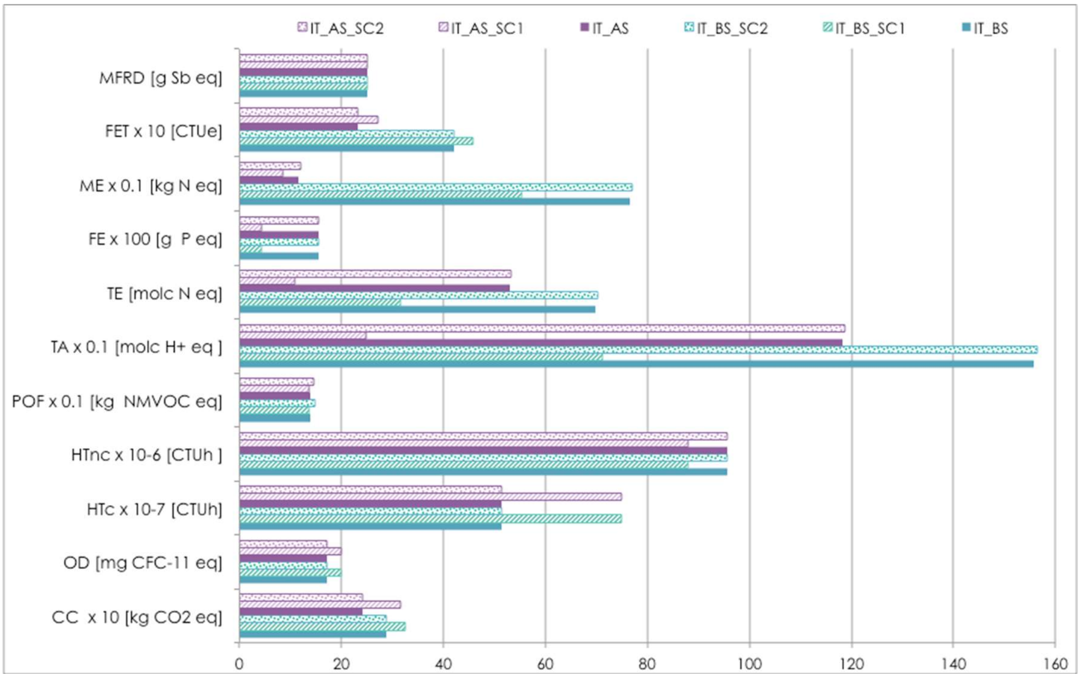


Figure 5 Results of the sensitivity analysis exploring type of fertilisers (SA1) and NOx emissions (SA2), for the Danish (top) and the Italian case studies (bottom) [Only affected impact categories are shown. Results are scaled to fit, factors in x-axis should be multiplied by values of each impact; impacts acronyms can be seen in section 2.4]

3.3 Comparison of the environmental impact of Barley production in Europe

In order to validate the results of this study, a comparison with available secondary data was performed, using datasets from Agrifootprint 1.0 (Blonk Agri-footprint BV, 2014). The study has considered barley production from the following countries: Germany (DE), France (FR), Belgium (BE), Ireland (IR) and United Kingdom (UK), considering economic allocation for co-products

Figure 6 shows that Italian and Danish systems are within the range of other European producers. The Italian barley production has the lowest CC with 265 kg CO_2 eq./t and the highest POF with 1.39 kg MNVOC eq., across all the countries considered. The Danish barley production shows the lowest impacts in seven out of 11 categories; specifically in HTCn (4.6 10-6 CTUh/t) and HTCnc (8.6 10-5 CTUh/t), AC (8.3 molc H^+ eq./t), TE (35.1 molc H^+ eq./t), FE (98.7 g P eq.), ME (3.35 Kg N eq./t) and FET (0.029 CTUe/t). Moreover, it also has the highest OD (19.2 mg CFC-11 eq./t) across all the countries.

Overall, only five impact categories estimated in this study are out of the range. In the case of OD and POF, the Italian and Danish barley systems show much higher values (4.6 and 1.9 times, respectively); the reason might be related to the agricultural field operations, the main contributors to these impacts. In the HTn, HTnc and FET, this study exhibits impacts much lower than the average European, between 3 to 26 times lower scores. Again, the reasons are related to the agricultural practices as field operations and agricultural inputs (fertilisers, pesticides and seeds use).

Finally, it is interesting to note that in the case of TA, TE and ME, the impacts are greatly affected by the modelling options (BS and AS), the scores move out from the range if different modelling options are applied. For instance, the highest impact scores, modelled by BS, are closer or within the range of the other European barley producers, however the lowest scores, calculated using AS, are lower than the range. This is especially true in the case of the Danish barley, which might support the theory of overestimation of certain emissions due to modelling options.

This analysis highlights how climate conditions, agriculture practices as well as emissions modelling are key issues when agriculture systems are assessed.

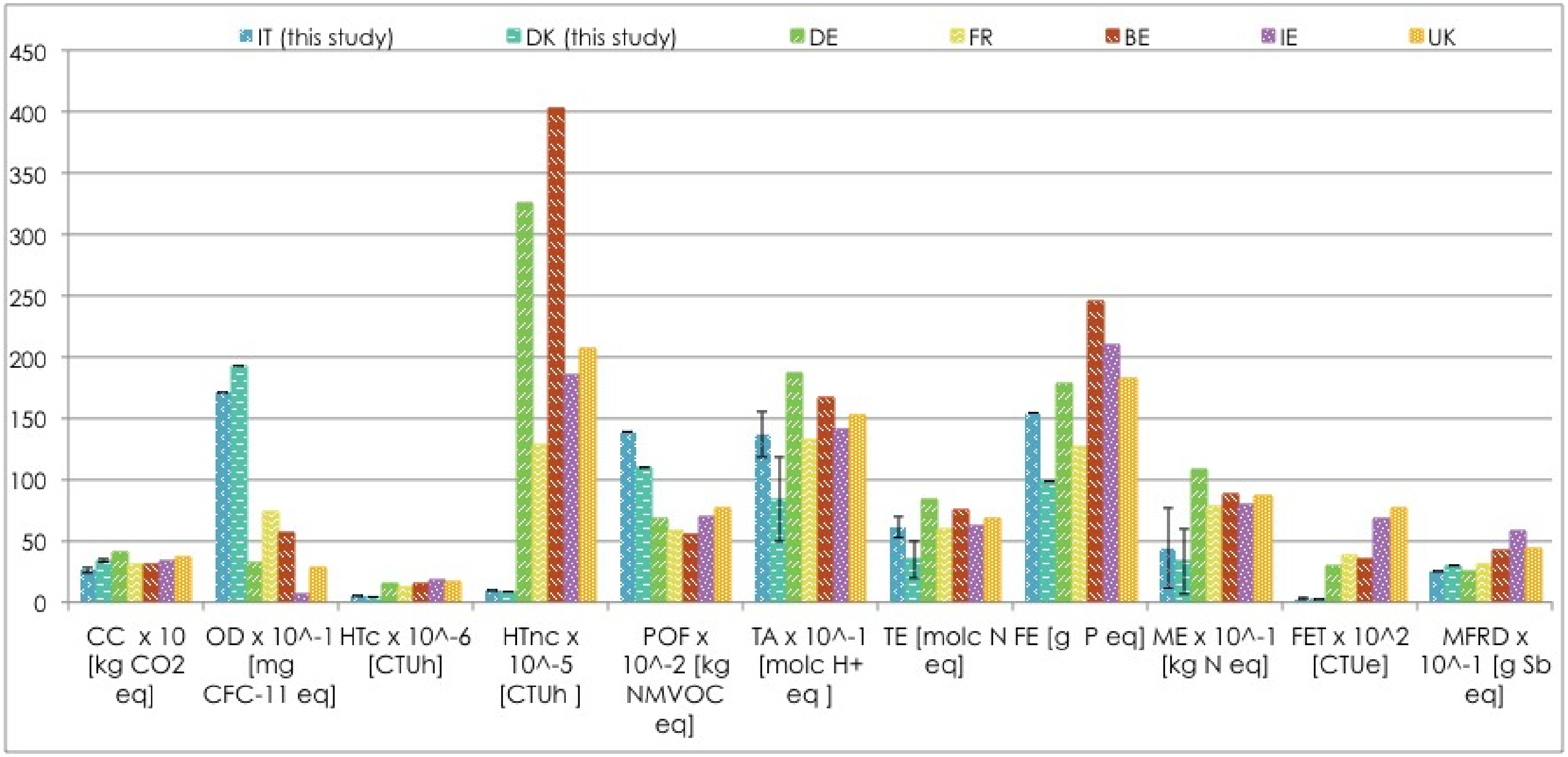


Figure 6 Comparison of the environmental of barley production between the current study and other European countries [All impacts express per 1 tonne of barley; the results from both modelling options (BS and AS) are displayed using error bars; DE: Germany, FR: France, BE: Belgium, IR: Ireland, UK: the UK; source Agrifootprint 1.0 (Blonk Agri-footprint BV, 2014)]

3.4 Recommendations for LCA practitioners

As discussed, the selection of the fertilisers and pesticides models does affect the environmental impacts estimated for agricultural systems, which in this case is Danish and Italian barley. In particular, fertilisers' emissions models greatly affect the environmental burden associated to CC, TA, TE and ME, as these categories strongly depend on nitrogen emissions. The impact on FET is instead affected by pesticides' emission and, consequently, by the selection of the emission models.

Through the case studies, it is seen the importance of the different agricultural practices and how they influence the results. The agricultural practices refer to the field operations carried out (type of machine, number of repletion and diesel fuel consumption), and the type and amount of fertilisers and pesticides used. All these factors are in part dependant on the pedo-climatic characteristics of the region of interest. Actions to reduce the environmental impact could involve, as shown in this study, the type of fertilisers applied and the optimization of their rate. However, as also seen in this study, the use of 100% mineral fertilisers would beneficiate some impact categories, in particular, those associated with nitrogen emissions, but also depending on the emission models, worsens some others. Therefore, the selection of the critical impacts will depend on the stakeholder's criteria, which would influence the management option toward impacts reduction.

To provide consumers and policy makers with reliable information on the environmental performances of food systems, LCA studies need to include all the relevant emissions outputs. In the case of pesticide emissions, consensus on the harmonization of LCI and LCIA modelling is going to be reached soon (Rosenbaum et al. 2015), meanwhile in the case of fertilisers, a consensus is still missing on a globally applicable model for calculating emissions. In the case of N emissions from fertiliser application, the main challenge is their quantification. Considering that primary data are usually unavailable, estimation tools are needed. In this regards, when different models are combined, the main issue is to prevent double counting of emissions. Guidance is therefore needed on which emissions should be included and which model should be applied to avoid double counting. Most of all studies on cereals include N_2O, NH_3, NOx and NO_3 emissions (Niero et al. 2015a,b, Fedele et al. 2014, Hamelin et al. 2012, Dijkman et al. 2016); meanwhile others do not consider the contribution from NOx, (Noya et al., 2015; Bacenetti et al., 2016b). Completeness in the LCI modelling of N emissions should be complemented by proper quantification of the impact pathways, for instance by the use of recent development in LCIA models for marine eutrophication (Cosme et al. 2015; Cosme and Niero, 2017; Cosme and Hauschild, 2016; Woods et al. 2016).

Concerning the impact related to pesticide application, LCA results should be complemented with other evaluation methods (e.g. risk assessment) before conclusions on corresponding risk levels for human health can be drawn.

Finally, this study can be used as a reliable source for other LCA studies, under Italian or Danish conditions, where barley is used as input to a broader LCA study e.i. beer, feed for food-producing animals or barley for human consumption, among many other products or systems.

4 Conclusions

This study has analysed the influence of the fertilisers and pesticides' emissions modelling options on the environmental impacts assessment of barley production, using as case studies Danish and Italian barley. The results show that the modelling options do affect the environmental burden of barley production, in particular CC, TA, TE, ME, FET, as these categories strongly depend on nitrogen and pesticides emissions. The most affected categories are ME, with variations up to 89%, and TA and TE, which vary around 60%. The smallest variations are seen in CC and FET, up to 15% and 45%, respectively. In the case of the scenario analyses, the main finding is that although including NOx emissions in the model does make LCA inventories more accurate, the results showed that on average the affected impact categories vary by ~7% maximum, except for POF which increases by up to 18%.

The use of models such as Brentrup (2000) and PESTLCI 2.0 (Dijkman et al. 2012) allows delivering more robust results and lowers overestimation of impacts, however, if site-specific data are not available, the conventional modelling practices as IPCC (2006) and Margni et al. (2002) still provide good estimates, in particular in widely used impacts as climate change.

Finally, to provide consumers and policy makers with reliable information on the environmental performances of food systems, LCA studies need to include all the relevant emissions outputs; therefore, up-to-date guidelines on how to properly account for N emissions from fertiliser use need to be developed to get consensus about fertiliser emissions quantification and modelling practices, as done for pesticides' emissions.

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6 References

AIC (2016). Associazione italiana coltivatori. Retried from http://www.aicnazionale.com/

Alhajj Ali, S., Tedone, L., Verdini, L., & De Mastro, G. (2017). Effect of different crop management systems on rainfed durum wheat greenhouse gas emissions and carbon footprint under Mediterranean conditions. *Journal of Cleaner Production, 140, Part 2*, 608-621. doi:http://dx.doi.org/10.1016/j.jclepro.2016.04.135

Anderson, P.M., E.A. Oelke, & S.R. Simmons (2013). Small Grains Production: Growth and development guide for spring barley. University of Minnesota, US. Retrieved from http://www.extension.umn.edu/agriculture/small-grains/growth-and-development/spring-barley/

Associazione Granaria di Milano (2016). Archivio quotazioni. Retrieved from http://www.granariamilano.org/

Audsley, E. (1997) Harmonisation of environmental life cycle assessment for agriculture. Final Report, Concerted Action AIR3-CT94-2028. European Commission, DG VI Agriculture, 139 pp.

Bacenetti J., Fusi A., Negri M., Guidetti R., Fiala M. (2014a). Environmental assessment of two different crop systems in terms of biomethane potential production. Science of the Total Environment, vol. 466-467, p. 1066-1077, ISSN: 0048-9697, doi: 10.1016/j.scitotenv.2013.07.109

Bacenetti J., Fusi A., Negri M., Fiala M. (2014b). Impact of cropping system and soil tillage on environmental performance of cereal silage productions. Journal of Cleaner Production, vol. 86, p. 49-59, ISSN: 0959-6526, doi: 10.1016/j.jclepro.2014.08.052

Bacenetti J., Duca D., Fusi A., Negri M., Fiala M. (2015). Mitigation strategies in the agro-food sector: the anaerobic digestion of tomato puree by-products. An Italian case study. Science of the Total Environment; vol. 526, p.88-97. doi.org/10.1016/j.scitotenv.2015.04.069

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. Science of the Total Environment, 539: 450-459. doi:10.1016/j.scitotenv.2015.09.015

Bacenetti, J., & Fusi, A. (2015). The environmental burdens of maize silage production: Influence of different ensiling techniques. Animal Feed Science and Technology, 204, 88-98. doi: http://dx.doi.org/10.1016/j.anifeedsci.2015.03.005

Bacenetti, J., Fusi, A., Negri, M., Bocchi, S., & Fiala, M. (2016b). Organic production systems: Sustainability assessment of rice in Italy. Agriculture, Ecosystems & Environment, 225, 33-44. doi: http://dx.doi.org/10.1016/j.agee.2016.03.046

Bacenetti J., Lovarelli D., Fiala M. (2016c). 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.

Baik, B.-K., & Ullrich, S. E. (2008). Barley for food: Characteristics, improvement, and renewed interest. Journal of Cereal Science, 48(2), 233-242. doi: http://dx.doi.org/10.1016/j.jcs.2008.02.002

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

Bannayan, M., Lakzian, A., Gorbanzadeh, N., & Roshani, A. (2011a). Variability of growing season indices in northeast of Iran. Theoretical Applied Climatology, 105, 485–494.

Bannayan, M., Lotfabadi, S., Sanjani, S., Mohammadian, A., & Agaalikhani, M. (2011b). Effects of precipitation and temperature on cereal yield variability in northeast of Iran. International Journal of Biometeorology, 55, 387–401.

Bannayan, M., & Sanjani, S. (2011). Weather conditions associated with irrigated crops in an arid and semi arid environment. Agricultural Forest Meteorology, 151, 1589–1598.

Baldoni R, Giardini L (2002a). Coltivazioni Erbacee: Cereali e Proteaginose (Herbaceous cultivations: cereals and oleaginous) Pàtron Editore (in Italian) Bologna: Italy.

Bartzas, G., Zaharaki, D., & Komnitsas, K. (2015). Life cycle assessment of open field and greenhouse cultivation of lettuce and barley. Information Processing in Agriculture, 2(3–4), 191-207. doi: http://dx.doi.org/10.1016/j.inpa.2015.10.001

Basset-Mens, C., Anibar, L., Durand, P., & Werf, H. M. G. (2006). Spatialised fate factors for nitrate in catchments: Modelling approach and implication for LCA results. Science of the Total Environment, 367, 367–382.

Behall, K. M., Scholfield, D. J., & Hallfrisch, J. G. (2006). Barley β-glucan reduces plasma glucose and insulin responses compared with resistant starch in men. Nutrition Research, 26(12), 644-650. doi: http://dx.doi.org/10.1016/j.nutres.2006.10.001

Birkved M, Hauschild MZ (2006) PestLCI: a model for estimating field emissions of pesticides in agricultural LCA. Ecol Model 198:433–451

Blonk Agri-footprint BV, 2014. Agri-Footprint. Description of Data. V1.0. PJ Gouda (The Netherlands)

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. Science of The Total Environment, Volume 553, 15 May 2016, Pages 551-564

Bonamente, E., Scrucca, F., Rinaldi, S., Merico, M. C., Asdrubali, F., & Lamastra, L. (2016). Environmental impact of an Italian wine bottle: Carbon and water footprint assessment. *Science of The Total Environment, 560–561*, 274-283. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.04.026

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. [journal article]. The International Journal of Life Cycle Assessment, 5(6), 349-357. doi: 10.1007/bf02978670

Bunzel, K., Schäfer, R. B., Thrän, D., & Kattwinkel, M. (2015). Pesticide runoff from energy crops: A threat to aquatic invertebrates?. Science of the Total Environment, 537, 187-196.

Buratti, C., Fantozzi, F., Barbanera, M., Lascaro, E., Chiorri, M., & Cecchini, L. (2017). Carbon footprint of conventional and organic beef production systems: An Italian case study. *Science of The Total Environment, 576*, 129-137. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.10.075

Capri, E., Karpouzas, D. 2007. Pesticide Risk Assessment in Rice Paddies: Theory and Practice. Elsevier, pp. 266.

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

Ciffroy, P., Alfonso, B., Altenpohl, A., Banjac, Z., Bierkens, J., Brochot, C., et al. (2016). Modelling the exposure to chemicals for risk assessment: a comprehensive library of multimedia and PBPK models for integration, prediction, uncertainty and sensitivity analysis–the MERLIN-Expo tool. Science of The Total Environment, 568, 770-784

Colomb, V., Amar, S.A., Basset Mens, C., Gac, A., Gaillard, G., Koch, P., Mousset, J., Salou, T., ATailleur, A., Hays, M.G. 2014. AGRIBALYSE®, the French LCI Database for agricultural products: high quality data for producers and environmental labelling. Oilseeds and fats, Crops and Lipids 22 (1).

Cóndor, R.D., Di Cristofaro, E., De Lauretis, R. (2008). Agricolture Inventario nazionale delle emissioni e disaggregazione provinciale. Istituto Superiore per la Protezione e la Ricerca Ambientale, ISPRA. Rapporto tecnico 85/2008. Roma, Italia (2008)

Cong, R.-G., & Termansen, M. (2016). A bio-economic analysis of a sustainable agricultural transition using green biorefinery. Science of The Total Environment, 571, 153-163. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.07.137

Cosme,N., Koskib, M., Hauschild, M. Z. 2015. Exposure factors for marine eutrophication impacts assessment based on a mechanistic biological model. Ecological Modelling. 317, 50–63. doi:10.1016/j.ecolmodel.2015.09.005

Cosme,N. & Hauschild, M.Z. 2016. Effect Factors for marine eutrophication in LCIA based on species sensitivity to hypoxia. Ecological Indicators. 69, 453–462. doi:10.1016/j.ecolind.2016.04.006

Cosme,N & Niero, M. 2017. Modelling the influence of changing climate in present and future marine eutrophication impacts from spring barley production. Journal of Cleaner Production 140, 537-546 http://dx.doi.org/10.1016/j.jclepro.2016.06.077

Dauriat, A.(2013). EuroBioRef. European Multilevel Integrated Biorefinery Design for Sustainable Biomass Processing. Biocore, EuroBioRef, Suprabio(2013), pp. 1–38

de Vries, W., Kros, J., Dolman, M. A., Vellinga, T. V., de Boer, H. C., Gerritsen, A. L., . . . Bouma, J. (2015). Environmental impacts of innovative dairy farming systems aiming at improved internal nutrient cycling: A multi-scale assessment. Science of The Total Environment, 536, 432-442. doi:http://dx.doi.org/10.1016/j.scitotenv.2015.07.079

Dijkman, T.J., Birkved, M., Hauschild, M.Z., 2012. PestLCI 2.0: a second generation model for estimating emissions of pesticides from arable land in LCA. Int. J. Life Cycle Assess. 17 (8), 973–986. doi: 10.1007/s11367-012-0439-2

Dijkman, T. J., Birkved, M., Saxe, H., Wenzel, H., & Hauschild, M. Z. (2017). Environmental impacts of barley cultivation under current and future climatic conditions. Journal of Cleaner Production 140, 644e653. doi: http://dx.doi.org/10.1016/j.jclepro.2016.05.154

Drastig, K., Prochnow, A., Libra, J., Koch, H., & Rolinski, S. (2016). Irrigation water demand of selected agricultural crops in Germany between 1902 and 2010. Science of The Total Environment, 569, 1299-1314.

DTU-MAN-QSA, (2016) Pesticide consensus in LCA. Technical University of Denmark (DTU), Department of Management Engineering (MAN), Division for Quantitative Sustainability Assessment (QSA), http://www.qsa.man.dtu.dk/dissemination/pesticide-consensus/pesticide-consensus-in-lca

EC (European Commission). Directive 2009/128/EC of the European parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides. Off J 2009;309:71. [24.11.2009].

EC (1999). The environmental impact of arable crop production in the European Union: practical options for improvement. Directorate-General XI of the Eurpean Commission.

Ecoil. (2006). Implementation of life cycle inventory in Ribera Baja (Navarra, Spain) http://www.ecoil.tuc.gr/Implementation%20of%20Life%20Cycle%20Inventory%20in%20Ribera%20Baja%20(Navarra,%20Spai).zip (2006) (Accessed 29 January 2011)

EFE-So (2015). Estimation of Fertilisers Emissions Software. Retrieved form http://www.sustainable-systems.org.uk/efeso.php

M. Emmenegger, J. Reinhard, R. Zah. (2009).Sustainability Quick Check for Biofuels — Intermediate Background Report Agroscope Reckenholz-Tänikon, Dübendorf (2009) (with contributions from Ziep, T., Weichbrodt, R., Wohlgemuth, V., FHTW Berlin, Roches, A. Freiermuth Knuchel, R., & Gaillard, G., 2009)

Eurostat (2016). http://appsso.eurostat.ec.europa.eu/nui/setupDownloads.do

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. Environ. Dev. Sustain. 14, 979–992. doi:10.1007/s10668-012-9367-3.

Fantin, V., Righi, S., Rondini, I., Masoni, P. (2017) Environmental assessment of wheat and maize production in an Italian farmers' cooperative. Journal of Cleaner Production. 140, 631-643. doi: http://dx.doi.org/10.1016/j.jclepro.2016.06.136

FAOSTAT (2016a). Production of Crops: barley. Retried from http://faostat3.fao.org/browse/Q/QC/E

FAOSTAT (2016b). Food and Agricultural commodities production: barley. Retried from http://faostat3.fao.org/browse/rankings/countries_by_commodity/E

Fedele, A., Mazzi, A., Niero, M., Zuliani, F., Scipioni A. (2014) Can the LCA methodology be adopted to support a single farm on its environmental impacts forecast evaluation between conventional and organic production? An Italian case study. Journal of Cleaner Production, 69, 49–59. http://dx.doi.org/10.1016/j.jclepro.2014.01.034

Figueiredo, F., Castanheira, É. G., & Freire, F. (2017). Life-cycle assessment of irrigated and rainfed sunflower addressing uncertainty and land use change scenarios. *Journal of Cleaner Production, 140, Part 2*, 436-444. doi:http://dx.doi.org/10.1016/j.jclepro.2016.06.151

Fusi A., Bacenetti J., González-García S., Vercesi A., Bocchi S., Fiala M. (2014). Environmental profile of paddy rice cultivation with different straw management. Science of the Total Environment, vol. 494-495, p. 119-128, ISSN: 0048-9697, doi: 10.1016/j.scitotenv.2014.06.126

Fusi, A., González-García, S., Moreira, M. T., Fiala, M., & Bacenetti, J. (2016). Rice fertilised with urban sewage sludge and possible mitigation strategies: an environmental assessment. Journal of Cleaner Production. doi: http://dx.doi.org/10.1016/j.jclepro.2016.04.089

Fusi, A., González-García, S., Moreira, M. T., Fiala, M., & Bacenetti, J. (2017). Rice fertilised with urban sewage sludge and possible mitigation strategies: an environmental assessment. Journal of Cleaner Production, 140, Part 2, 914-923. doi:http://dx.doi.org/10.1016/j.jclepro.2016.04.089

Gallejones, P., Pardo, G., Aizpurua, A., & del Prado, A. (2015). Life cycle assessment of first-generation biofuels using a nitrogen crop model. Science of The Total Environment, 505, 1191-1201.doi:http://dx.doi.org/10.1016/j.scitotenv.2014.10.061

Goglio, P., Grant, B. B., Smith, W. N., Desjardins, R. L., Worth, D. E., Zentner, R., & Malhi, S. S. (2014). Impact of management strategies on the global warming potential at the cropping system level. Science of the Total Environment, 490, 921-933. doi.org/10.1016/j.scitotenv.2014.05.070.

González-García, S., Baucells, F., Feijoo, G., & Moreira, M. T. (2016). Environmental performance of sorghum, barley and oat silage production for livestock feed using life cycle assessment. Resources, Conservation and Recycling, 111, 28-41. doi: http://dx.doi.org/10.1016/j.resconrec.2016.04.002

Hamelin, L., Jørgensen, U., Petersen, B.M., Olesen, J.E.,Wenzel, H., 2012. Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: a consequential life cycle inventory. GCB Bioenergy 4, 889–907. doi:10.1111/j.1757-1707.2012.01174.x.

Hauggaard-Nielsen, H., Lachouani, P., Knudsen, M. T., Ambus, P., Boelt, B., & Gislum, R. (2016). Productivity and carbon footprint of perennial grass–forage legume intercropping strategies with high or low nitrogen fertilizer input. Science of The Total Environment, 541, 1339-1347. doi:http://dx.doi.org/10.1016/j.scitotenv.2015.10.013

Hauschild, M.Z., Goedkoop, M., Guin.e, J., Heijungs, R., Huijbregts, M., Jolliet, O., et al.,2013. Identifying best existing practice for characterization modeling in life cycle impact assessment. Int. J. Life Cycle Assess. 18, 683–697. doi:10.1007/s11367-012-0489-5.

Hayashi, K., Nagumo, Y., & Domoto, A. (2016). Linking environment-productivity trade-offs and correlated uncertainties: Greenhouse gas emissions and crop productivity in paddy rice production systems. Science of The Total Environment, 571, 134-141. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.07.138

Hendy, C. R. C., Kleih, U., Crawshaw, R., and Phillips, M. (1995). Livestock and the Environment Finding a Balance. Interactions between Livestock Production Systems and the Environment, Impact Domain: concentrate feed demand. NATURAL RESOURCES INSTITUTE, UK. Retrieved from http://www.fao.org/wairdocs/lead/x6123e/x6123e00.htm#Contents

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

ISO, 2006a. ISO 14040:2006. Environmental Management. Life cycle assessment. Principle and Framework. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006b. ISO 14044:2006. Environmental Management. Life cycle assessment. Requirements and Guidelines. International Organization for Standardization, Geneva, Switzerland.

Jones, H., Civáň, P., Cockram, J., Leigh, F. J., Smith, L. M., Jones, M. K., Charles, M. P., Molina-Cano, J.L., Powell, W., Jones, G., Brown, T. A. (2011). Evolutionary history of barley cultivation in Europe revealed by genetic analysis of extant landraces. [journal article]. BMC Evolutionary Biology, 11(1), 1-12. doi: 10.1186/1471-2148-11-320

Korsaeth, A., Henriksen, T. M., Roer, A.-G., & Hammer Strømman, A. (2014). Effects of regional variation in climate and SOC decay on global warming potential and eutrophication attributable to cereal production in Norway. Agricultural Systems, 127, 9-18. doi:http://dx.doi.org/10.1016/j.agsy.2013.12.007

Khoshnevisan, B., Rajaeifar, M.a., Clark, S., Shamahirband, S., Anuar, N.B., Mohd Shuib, , N.L., Gani, A. (2014). Evaluation of traditional and consolidated rice farms in Guilan Province, Iran, using life cycle assessment and fuzzy modelling. Science of The Total Environment, Volume 481, 15 May 2014, Pages 242-251

Laratte, B., Guillaume, B., Kim, J., Birregah, B. (2014). Modeling cumulative effects in life cycle assessment: The case of fertilizer in wheat production contributing to the global warming potential. Science of The Total Environment, Volume 481, 15 May 2014, Pages 588-595

Lechon, Y., Cabal, H. and Saez, R., 2005. Life cycle analysis of wheat and barley crops for bioethanol production in Spain. International journal of agricultural resources, governance and ecology, 4(2), pp.113-122.

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 from anaerobic co-digestion of pig slurry and energy crops. Renewable Energy, vol. 68, p. 625-635, doi: 10.1016/j.renene.2014.03.005

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 Management 41, 50-59. DOI: 10.1016/j.wasman.2015.03.043

Margni M., Jolliet O., Crettaz P. (2002) Life cycle assessment of pesticides on human health and ecosystems. Agriculture, Ecosystems & Environment 93: 379– 392

Meneses, M., Torres, C. M., & Castells, F. (2016). Sensitivity analysis in a life cycle assessment of an aged red wine production from Catalonia, Spain. Science of The Total Environment, 562, 571-579. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.04.083

Milà i Canals, L. (2003). Contributions to LCA Methodology for Agricultural Systems. (Disseration) Universitat Autònoma de Barcelona (2003)

Muñoz-Amatriaín, M., Cuesta-Marcos, A., Endelman, J. B., Comadran, J., Bonman, J. M., Bockelman, H. E., Chao,S., Russell, J., Waugh, R., Hayes, P.M., Muehlbauer, G. J. (2014). The USDA Barley Core Collection: Genetic Diversity, Population Structure, and Potential for Genome-Wide Association Studies. PLoS ONE, 9(4), e94688. doi: 10.1371/journal.pone.0094688.

Nemecek T. & Kägi T., 2007. Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent V2.0 No. 15. Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, CH. Retrieved from: www.ecoinvent.ch.

Nemecek T., BengoaX.,Lansche J.,MouronP., Rossi V. & Humbert S. (2014) Methodological Guidelines for th Life Cycle Inventory of AgriculturalProducts Version 2.0, July 2014. World Food LCA Database (WFLDB). Quantis and Agroscope, Lausanne and Zurich, Switzerland.

Niero, M., Ingvordsen, C.H., Jørgensen, R.B., Hauschild, M.Z. (2015a) How to manage uncertainty in future Life Cycle Assessment (LCA) scenarios addressing the effect of climate change in crop production. J. Clean. Prod. 107, 693e706. http://dx.doi.org/10.1016/j.jclepro.2015.05.061.

Niero, M., Ingvordsen, C. H., Peltonen-Sainio, P., Jalli, M., Lyngkjær, M. F., Hauschild, M. Z., & Jørgensen, R. B. (2015b). 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. doi: http://dx.doi.org/10.1016/j.agsy.2015.02.007

Nikkhah, A., Emadi, B., Hamzeh, S., Firouzi, S., Rosentrater, K.A., Allahyari, M.A. (2016) Integration of life cycle assessment and Cobb-Douglas modeling for the environmental assessment of kiwifruit in Iran. Journal of Cleaner Production, 2016, 137, 843-849. doi:10.1016/j.jclepro.2016.07.151

Nikkhah, A., Khojastehpour, M., Emadi, B., Taheri-Rad, A., Khorramdel, S. (2015) Environmental impacts of peanut production system using life cycle assessment methodology, Journal of Cleaner Production, 2015, 92, 84-90. Doi:10.1016/j.jclepro.2014.12.048

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. Journal of Cleaner Production 140; 399-409 doi: http://dx.doi.org/10.1016/j.jclepro.2016.06.071

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). Journal of Cleaner Production, vol. 2015, p. 1-16, ISSN: 0959-6526, doi: 10.1016/j.jclepro.2015.03.001

Oldfield, T. L., Achmon, Y., Perano, K. M., Dahlquist-Willard, R. M., VanderGheynst, J. S., Stapleton, J. J., . . . Holden, N. M. (2017). A life cycle assessment of biosolarization as a valorization pathway for tomato pomace utilization in California. Journal of Cleaner Production, 141, 146-156. doi:http://dx.doi.org/10.1016/j.jclepro.2016.09.051

Peter, C., Fiore, A., Hagemann, U., Nendel, C., Xiloyannis, C, 2016. Improving the accounting of field emissions in the carbon footprint of agricultural products: a comparison of default IPCC methods with readily available medium-effort modeling approaches. International Journal of Life Cycle Assessment (2016) 21: 791. doi:10.1007/s11367-016-1056-2.

Pirlo, G., Carè, S., Casa, G. D., Marchetti, R., Ponzoni, G., Faeti, V., . . . Falconi, F. (2016). Environmental impact of heavy pig production in a sample of Italian farms. A cradle to farm-gate analysis. *Science of The Total Environment, 565*, 576-585. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.04.174Pluimers, J., 2001. An Environmental Systems Analysis of Greenhouse Horticulture in the Netherlands. Ph.D. thesis, Wageningen University, The Netherlands.

Poritosh. R, N. Daisuke, O. Takahiro, X. Qingyi, O. Hiroshi, N. Nobutaka, S. Takeo. A review of life cycle assessment (LCA) on some food products. Journal of Food Engineering, 90 (2009), p. 1

Pre Consultants. 2015. SimaPro Life Cycle Analysis version 7.2 (software).

Pryor, S. W., Smithers, J., Lyne, P., & van Antwerpen, R. (2017). Impact of agricultural practices on energy use and greenhouse gas emissions for South African sugarcane production. Journal of Cleaner Production, 141, 137-145. doi:http://dx.doi.org/10.1016/j.jclepro.2016.09.069

Rafiee, S., Khoshnevisan, B., Mohammadi, I., Aghbashlo, M., mousazadeh, H., & Clark, S. (2016). Sustainability evaluation of pasteurized milk production with a life cycle assessment approach: An Iranian case study. *Science of The Total Environment, 562*, 614-627. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.04.070

Rakotovao, N. H., Razafimbelo, T. M., Rakotosamimanana, S., Randrianasolo, Z., Randriamalala, J. R., & Albrecht, A. (2017). Carbon footprint of smallholder farms in Central Madagascar: The integration of agroecological practices. *Journal of Cleaner Production, 140, Part 3*, 1165-1175. doi:http://dx.doi.org/10.1016/j.jclepro.2016.10.045

Rana, R., Ingrao, C., Lombardi, M., & Tricase, C. (2016). Greenhouse gas emissions of an agro-biogas energy system: Estimation under the Renewable Energy Directive. *Science of The Total Environment, 550*, 1182-1195. doi:http://dx.doi.org/10.1016/j.scitotenv.2015.10.164

Remoundou, K., et al. (2015). Perceptions of pesticides exposure risks by operators, workers, residents and bystanders in Greece, Italy and the UK. Science of the Total Environment, 505: 1082-1092.

Renzulli, P. A., Bacenetti, J., Benedetto, G., Fusi, A., Ioppolo, G., Niero, M., Supino, S. (2015). Life Cycle Assessment in the Cereal and Derived Products Sector. In B.

Notarnicola, P. A. Renzulli, R. Salomone, R. Roma, L. Petti, & A. K. Cerutti (Eds.), Life Cycle Assessment in the Agri-food Sector: Case Studies, Methodological Issues and Best Practices. (pp. 185-250). Chapter 4.Springer. DOI: 10.1007/978-3-319-11940-3_4

Roer, A.-G., Korsaeth, A., Henriksen, T. M., Michelsen, O., & Strømman, A. H. (2012). The influence of system boundaries on life cycle assessment of grain production in central southeast Norway. Agricultural Systems, 111, 75-84. doi:http://dx.doi.org/10.1016/j.agsy.2012.05.007

Rosenbaum, R. K., Anton, A., Bengoa, X., Bjørn, A., Brain, R., Bulle, C., . . . Wallman, M. (2015). The Glasgow consensus on the delineation between pesticide emission inventory and impact assessment for LCA. [journal article]. The International Journal of Life Cycle Assessment, 20(6), 765-776. doi: 10.1007/s11367-015-0871-1

Suciu, N., Tediosi, A., Ciffroy, P., Altenpohl, A., Brochot, C., Verdonck, F., Ferrari, F., Giubilato, E., Capri, E., Fait, G. (2016). Potential for MERLIN-Expo, an advanced tool for higher tier exposure assessment, within the EU chemical legislative frameworks. Science of The Total Environment, 562, 474-479.

Sullivan, P., Arendt, E., & Gallagher, E. (2013). The increasing use of barley and barley by-products in the production of healthier baked goods. Trends in Food Science & Technology, 29(2), 124-134. doi: http://dx.doi.org/10.1016/j.tifs.2012.10.005

Tecco, N., Baudino, C., Girgenti, V., & Peano, C. (2016). Innovation strategies in a fruit growers association impacts assessment by using combined LCA and s-LCA methodologies. Science of The Total Environment, 568, 253-262. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.05.203

Vagnoni, E., Franca, A., Breedveld, L., Porqueddu, C., Ferrara, R., & Duce, P. (2015). Environmental performances of Sardinian dairy sheep production systems at different input levels. Science of The Total Environment, 502, 354-361. doi:http://dx.doi.org/10.1016/j.scitotenv.2014.09.020

Wandl, M.-T., & Haberl, H. (2017). Greenhouse gas emissions of small scale ornamental plant production in Austria - A case study. *Journal of Cleaner Production, 141*, 1123-1133. doi:http://dx.doi.org/10.1016/j.jclepro.2016.09.093

Wang, Z.-b., Chen, J., Mao, S.-c., Han, Y.-c., Chen, F., Zhang, L.-f., . . . Li, C.-d. (2017). Comparison of greenhouse gas emissions of chemical fertilizer types in China's crop production. *Journal of Cleaner Production, 141*, 1267-1274. doi:http://dx.doi.org/10.1016/j.jclepro.2016.09.120

Woods, J.S., Veltman, K., Huijbregtsc, M.A.J., Veronesa, F., Hertwichd, E.G. 2016. Towards a meaningful assessment of marine ecological impacts in life cycle assessment (LCA). Environment Interantional, 89–90, 48–61. doi:10.1016/j.envint.2015.12.033

XE (2016). Convertion rate. Retrieved from www.xe.com

Xia, L., Ti, C., Li, B., Xia, Y., & Yan, X. (2016). Greenhouse gas emissions and reactive nitrogen releases during the life-cycles of staple food production in China and their mitigation potential. Science of The Total Environment, 556, 116-125. doi:http://dx.doi.org/10.1016/j.scitotenv.2016.02.204