Journal of Environmental Management 149 (2015) 193–208

Contents lists available at ScienceDirect

Journal of Environmental Management

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

Review

Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment?

Matthias S. Meier [a, *], Franziska Stoessel [b], Niels Jungbluth [c], Ronnie Juraske [b], Christian Schader [a], Matthias Stolze [a]

[a] FiBL – Research Institute of Organic Agriculture, Ackerstrasse 113, P.O. Box 219, 5070 Frick, Switzerland
[b] ETH Zurich – Institute of Environmental Engineering, John-von-Neumann-Weg 9, CH-8093 Zurich, Switzerland
[c] ESU-services Ltd., Margrit Rainer-Strasse 11c, 8050 Zurich, Switzerland

A R T I C L E   I N F O

Article history:
Received 23 August 2013
Received in revised form
12 August 2014
Accepted 9 October 2014
Available online

Keywords:
Comparative life cycle assessment (LCA)
Organic
Conventional
Agriculture
Attributional
Consequential
N-surplus

A B S T R A C T

Comprehensive assessment tools are needed that reliably describe environmental impacts of different agricultural systems in order to develop sustainable high yielding agricultural production systems with minimal impacts on the environment. Today, Life Cycle Assessment (LCA) is increasingly used to assess and compare the environmental sustainability of agricultural products from conventional and organic agriculture. However, LCA studies comparing agricultural products from conventional and organic farming systems report a wide variation in the resource efficiency of products from these systems. The studies show that impacts per area farmed land are usually less in organic systems, but related to the quantity produced impacts are often higher. We reviewed 34 comparative LCA studies of organic and conventional agricultural products to analyze whether this result is solely due to the usually lower yields in organic systems or also due to inaccurate modeling within LCA. Comparative LCAs on agricultural products from organic and conventional farming systems often do not adequately differentiate the specific characteristics of the respective farming system in the goal and scope definition and in the inventory analysis. Further, often only a limited number of impact categories are assessed within the impact assessment not allowing for a comprehensive environmental assessment. The most critical points we identified relate to the nitrogen (N) fluxes influencing acidification, eutrophication, and global warming potential, and biodiversity. Usually, N-emissions in LCA inventories of agricultural products are based on model calculations. Modeled N-emissions often do not correspond with the actual amount of N left in the system that may result in potential emissions. Reasons for this may be that N-models are not well adapted to the mode of action of organic fertilizers and that N-emission models often are built on assumptions from conventional agriculture leading to even greater deviances for organic systems between the amount of N calculated by emission models and the actual amount of N available for emissions. Improvements are needed regarding a more precise differentiation between farming systems and regarding the development of N emission models that better represent actual N-fluxes within different systems. We recommend adjusting N- and C-emissions during farmyard manure management and farmyard manure fertilization in plant production to the feed ration provided in the animal production of the respective farming system leading to different N- and C-compositions within the excrement. In the future, more representative background data on organic farming systems (e.g. N content of farmyard manure) should be generated and compiled so as to be available for use within LCA inventories. Finally, we recommend conducting consequential LCA – if possible – when using LCA for policy-making or strategic environmental planning to account for different functions of the analyzed farming systems.

1. Introduction

Agriculture's impacts on the environment are substantial (Foley et al., 2005, 2011). In particular modern agriculture is accelerating the rate of biodiversity loss and is one of the major drivers of climate change and human induced changes to the nitrogen cycle,

* Corresponding author. Tel.: +41 62 865 72 20; fax: +41 62 865 72 73.
E-mail addresses: matthias.meier@fibl.org (M.S. Meier), franziska.stoessel@ifu.baug.ethz.ch (F. Stoessel), jungbluth@esu-services.ch (N. Jungbluth), juraske@ifu.baug.ethz.ch (R. Juraske), christian.schader@fibl.org (C. Schader), matthias.stolze@fibl.org (M. Stolze).

http://dx.doi.org/10.1016/j.jenvman.2014.10.006

with these three processes having already exceeded the Earth's boundaries (Rockström et al., 2009). In order to become more sustainable farming systems should be developed and applied that minimize externalities by optimizing the use of internal production inputs (e.g. of farmyard manure) (Nemecek et al., 2011b) and/or implement ecological intensification, which involves replacing external inputs with ecosystem services (e.g. by enhancing natural biocontrol) while maintaining or even increasing yield levels (Bommarco et al., 2013).

Organic farming is often proposed as solution to reduce agriculture's impacts on the environment (Seufert et al., 2012b). However, yields in organic agriculture are usually lower than in conventional agriculture. For example, crop yield differences between organic and conventional systems range – while strongly depending on system and site characteristics – from 5 to 34% (de Ponti et al., 2012; Seufert et al., 2012a). So, more land is usually required to produce the same amount of food in organic farming systems than in conventional farming. Thus, the environmental benefits per product unit of organic farming might be outweighed; as was argued in the recent meta-analysis by Tuomisto et al. (2012).

In order to develop more sustainable farming systems, researchers and decision-makers need information about the strengths and weaknesses of different farming systems with respect to productivity and environmental impacts within the ecosystems' carrying capacity. Therefore, assessment tools are required that allow for comprehensive environmental impact assessments of different farming systems to enable informed conclusions.

Life cycle assessment (LCA) is increasingly used to assess the ecological sustainability of food products and is seen as a useful tool to evaluate environmental impacts of food products and production systems (Roy et al., 2009). LCA is the most comprehensive method available and useful for avoiding problem-shifting e.g., from one phase of the life cycle to another because it analyzes potential environmental impacts throughout a product's life cycle (ISO, 2006) including the supply chain and downstream processes (Finnveden et al., 2009). Results from LCAs may form the basis for making decisions for policy makers, producers as well as for consumers in selecting sustainable products and production processes (Roy et al., 2009).

A growing number of LCA studies has compared the environmental impacts of the same products produced in organic vs. conventional agriculture (see Table 1). Most of these LCA studies have found a lower environmental burden from organically produced products on a per area and year basis, but higher impacts have been found when evaluating emissions per product unit (e.g. Nemecek et al. (2011a) and the studies reviewed therein). Lower yields of organic farming systems leading to higher environmental impacts on a per product basis are seen as their main drawback (Tuomisto et al., 2012).

However, contemporary LCA studies report a wide variation in the resource efficiency of products from organic and conventional agriculture (e.g. studies on milk by Cederberg and Mattsson (2000), Williams et al. (2006), Thomassen et al. (2008b), van der Werf et al. (2009)). Some of this variation may be explained by yield differences between organic and conventional agriculture, while some of the variation may depend more on farmer's management choices than on the farming system itself (Tuomisto et al., 2012). Alternatively, some of the variation reported by comparative LCAs of products from different farming systems may also be due to inaccurate modeling of characteristics specific to the farming systems related to the assessed products.

The objectives of this review are:

a) to determine the parameters leading to differences in environmental impacts between organic and conventional products within comparative LCAs; and
b) to analyze, whether these parameters reflected farming system specific differences adequately.

Further, we analyze whether comparative LCA studies on organic and conventional products can be used to draw general conclusions on the environmental performance of organic and conventional farming systems. Finally, the objective is to show how LCA can be improved to better differentiate between products from different farming systems.

2. Methods

2.1. Review of peer-reviewed comparative LCA studies and LCA study reports

2.1.1. Literature search

We searched the ISI Web of Knowledge literature database (www.isiwebofknowledge.com) and the Scopus database (www.scopus.com) for LCA studies that compared organically and conventionally (i.e. non organic) produced commodities with no restriction on publication year or geographical context although review articles were excluded from the analysis. The search string "life cycle assessment AND organic AND conventional" was used in combination with different keywords including milk, beef, pig, poultry, arable crops, fruits and vegetables. In peer reviewed journals and conference proceedings, we found 31 comparative LCA studies and studies using LCA methodology to assess only a single impact category (e.g. carbon footprint studies). Since we searched academic literature databases, this review includes only studies which primarily aimed at answering academic questions. However, such studies may serve as the scientific basis for decision making, such as on a regulatory level.

In addition we included three scientific reports, which were available on the internet, on comparative LCAs from the UK (Williams et al., 2006), Sweden (Cederberg and Flysjö, 2004), and Switzerland (Alig et al., 2012). These three reports were not peer reviewed although they are well known within the LCA community dealing with food and agriculture. The report from Sweden was the basis for the peer reviewed study by Flysjö et al. (2012) and the report from the UK was the basis for the peer reviewed study by Williams et al. (2010). Both peer reviewed studies were also included in this review. All of the 34 studies that were reviewed are listed in Table 1, which also indicates the commodities, the country, and the underlying data basis.

Further, we added inventories on organic and conventional products from ecoinvent v2.2 and from ESU-services Ltd. (Jungbluth et al., 2013) to the studies found in literature and included them in our analyses (see Section 2.2).

2.1.2. Evaluation criteria

The main focus of this review of LCA studies and inventories is on the question of how organic and conventional farming systems were differentiated and modeled within comparative LCAs in order to assess and compare environmental impacts of agricultural food products. The review was guided by the following evaluation criteria:

1. Goal and scope definition
   - What was the goal of the LCA?
   - Was the LCA conducted with an attributional or consequential perspective?
   - What allocation rules were applied?

- What system boundaries were chosen?
- What functional units were used?

2. Inventory
   - What was the data basis used (experimental data vs. modeled data)?
   - What assumptions were taken regarding farming practices (including yields)?
   - What emission calculation models were used?
   - Were site-specific emission and characterization factors applied?
3. Impact assessment
   - Which impact categories were assessed?
   - Which life cycle impact assessment (LCIA-) methods were used?
4. Interpretation of results
   - Were sensitivity analyses to choices of methods conducted?
   - Were uncertainty analyses of results conducted?
   - What conclusions were drawn?

2.1.3. Analysis of studies

The studies were grouped according to the commodities that were analyzed and each study was analyzed according to the evaluation criteria listed above (see Supplementary Material). If not explicitly reported in the studies, we calculated the environmental impacts per unit of area and year additionally to the impacts reported per unit of product. This way the cultivation intensity, and how impacts related to the different agricultural systems before dividing by the amount of yield, became transparent. In the studies of Kavargiris et al. (2009), Litskas et al. (2011), Michos et al. (2012), and Zafiriou et al. (2012) impacts were reported per area only. For these studies, we calculated impacts per product based on the yields reported in these studies. Furthermore, the productivity as the amount of product per area was calculated if it was not explicitly stated in a paper. The relative differences between the impacts and yields of organic and conventional farming systems were calculated for each study (see also Supplementary Material).

In some studies, organic farming practices were compared with several conventional systems of different intensities (Haas et al., 2001; Cederberg and Flysjö, 2004; Casey and Holden, 2006; Williams et al., 2006; Warner et al., 2010; Nemecek et al., 2011a; Alig et al., 2012; Leinonen et al., 2012a,b; Michos et al., 2012; Zafiriou et al., 2012; Abeliotis et al., 2013; Villanueva-Rey et al., 2014). To analyze how products from farming systems that differ substantially are assessed in LCA, we considered only the comparisons between organic agriculture with the highest intensity levels of conventional agriculture. A range of variants, including low-, upland, and alpine production systems in milk (Hörtenhuber et al., 2010) and suckler cow and feedlot systems in beef production (Alig et al., 2012) within organic and conventional agriculture were analyzed to identify differences between the environmental impacts of organic and conventional agriculture for each of the variants. No comparisons were carried out across variants. Some studies included transportation, storage, and/or processing after the farm gate (Grönroos et al., 2006; Meisterling et al., 2009; Liu et al., 2010; Alig et al., 2012). However, since the systemic differences between organic and conventional farming occur within agricultural production, we only considered the agricultural production phase in our analyses (cradle-to-farm gate). Further quantitative and qualitative data were extracted wherever possible such as to compare surplus nitrogen with the amount of nitrogen from the emissions' modeling.

2.2. Analysis of inventory data

We supplemented the overview of published environmental impacts for organic and conventional products (Table 2) with inventory data from ESU-services Ltd. (Jungbluth et al., 2013) on milk, beef, pork, poultry, tomatoes, carrots, strawberries and pears; and ecoinvent inventories v2.2 on wheat, barley, soybean, and potatoes (Table 3): all of which are available for Swiss organic and integrated production (IP) (Nemecek et al., 2007). IP production in this paper refers to the definition in Nemecek et al. (2011a) including principles such as equilibrated nutrient balance, ecological compensation areas, diversified crop rotation, soil protection during winter to reduce the risk of erosion and nitrate leaching, and targeted and restricted application of pesticides. For this overview, we considered only impacts reported per unit of product (Table 3). Ecoinvent inventories are widely used as background data in LCA studies so critical points on the inventory level regarding emissions' modeling are, therefore, potentially translated to any respective LCA study that uses these inventory data.

3. Results and discussion

3.1. Overview of studies reviewed

3.1.1. Scope of the studies

In total, 34 studies that used LCA methodology and which compared milk, beef, pork, poultry, eggs, fruits, vegetables, nuts, and arable crops from organic and conventional agriculture were reviewed (Table 1). Some studies compared more than one product (Grönroos et al., 2006; Williams et al., 2006, 2010; Bos et al., 2007; Nemecek et al., 2011a; Alig et al., 2012; Venkat, 2012). Milk was the product most often compared between organic and conventional agriculture (11 out of 34 studies reviewed), while six studies dealt with meat from different production systems, one study analyzed egg production and 19 studies compared various plant products. All but four studies (Cederberg and Flysjö, 2004; Williams et al., 2006, 2010; Flysjö et al., 2012) were fully independent and mostly used data from real farms to assess the environmental impacts of products from different production systems (Table 1). In Flysjö et al. (2012), the farm inventories from Cederberg and Flysjö (2004) were used for further analyses and Williams et al. (2010) built upon Williams et al. (2006).

In the cases of milk, beef, pig, and egg production, all of the reviewed studies refer to middle or northern European agriculture (Table 1). In the case of poultry, one study was conducted in southern Europe in addition to two studies from middle and northern Europe. Of the studies on fruit, vegetables and arable crops, one study analyzed pear (Liu et al., 2010) and one soybean production systems in China (Knudsen et al., 2010). Further, two studies on different crops were conducted in the USA (Meisterling et al., 2009; Venkat, 2012). All of the other studies on fruits, vegetables, and arable crops were conducted in the context of European agriculture.

Almost all of the reviewed studies compared organic with conventional production systems to elicit which farming system is the most environmentally sustainable for the analyzed products. Seven studies furthermore aimed at identifying hot-spots of environmental impacts to enable deduction of mitigation options to reduce environmental impacts of farming systems (Cederberg and Mattsson, 2000; Basset-Mens and van der Werf, 2005; Grönroos et al., 2006; van der Werf et al., 2009; Hörtenhuber et al., 2010; Alig et al., 2012; Guerci et al., 2013). Abeliotis et al. (2013) further analyzed the environmental impacts of using different bean varieties. Meisterling et al. (2009) also compared agricultural impacts on global warming potential (GWP) with transport impacts. Venkat (2012), in addition, analyzed the scenario of converting production of the analyzed products from conventional to organic estimating the potential for sequestering additional organic carbon in the soil. Finally, one study used data from organic and conventional milk

Table 1
Comparative LCA studies reviewed.

Study	Products analyzed	Country	Data basis
Abeliotis et al. (2013)	Bean	Greece	Several producers involved in a labeling schemes (to derive average agricultural practice in the region under study)
Alig et al. (2012)	Beef, pig, poultry	Switzerland	Beef: 14 model farms based on data from 2534 conventional/1818 organic farms; Pig: 6 model farms based on data from 5397 conventional/258 organic farms Poultry: 3 production scenarios based on production data from one meat processing company
Basset-Mens and van der Werf (2005)	Pig	France	3 production scenarios based on French official farm statistical data and expert judgment, data from one local feed producer
de Backer et al. (2009)	Leek	Belgium	1 organic/1 conventional agricultural research institute
Boggia et al. (2010)	Poultry	Italy	1 organic/1 conventional farm
Bos et al. (2007)	Potato, sugar beet, pea, leek, lettuce, beans	The Netherlands	Model farms for different farm types, data origin not further specified
Casey and Holden (2006)	Beef	Ireland	5 organic/5 conventional farms
Cederberg and Mattsson (2000)	Milk	Sweden	1 organic/1 conventional farm
Cederberg and Flysjö (2004)	Milk	Sweden	6 organic/9 conventional farms
Flysjö et al. (2012)	Milk	Sweden	6 organic/9 conventional farms
Grönroos et al. (2006)	Milk, rye	Finland	1 organic/1 conventional farm
Guerci et al. (2013)	Milk	Denmark	2 organic/3 conventional farms
Haas et al. (2001)	Milk	Germany	6 organic/6 conventional
Hörtenhuber et al. (2010)	Milk	Austria	Official Austrian farm statistical data (IACS database)
Juraske and Sanjuán (2011)	Orange	Spain	Typical production conditions from a Spanish orange production region
Kavargiris et al. (2009)	Grape	Greece	9 organic/9 conventional farms
Knudsen et al. (2010)	Soybean	China	20 organic/15 conventional farms
Kristensen et al. (2011)	Milk	Denmark	32 organic/35 conventional farms
Leinonen et al. (2012a)	Poultry	UK	Industry data/national inventories/database data
Leinonen et al. (2012b)	Eggs	UK	Industry data/national inventories/database data
Litskas et al. (2011)	Cherry	Greece	10 organic/10 conventional orchards
Liu et al. (2010)	Pear	China	3 organic/2 conventional farms
Meisterling et al. (2009)	Wheat	USA	Farm statistical data/literature data
Michos et al. (2012)	Peach	Greece	3 organic/4 conventional farms
Nemecek et al. (2011a)	2 crop rotations of arable crops	Switzerland	Long term field trials
Thomassen et al. (2008b)	Milk	Netherlands	11 organic/10 conventional farms
van der Werf et al. (2009)	Milk	France	6 organic/41 conventional farms
Venkat (2012)	Alfalafa, blueberry, apple, wine grape, raisin grape, strawberry, almond, walnut, broccoli, lettuce	USA (California)	Literature data (cost and return studies)
Vermeulen and van der Lans (2011)	Tomato	Netherlands	Statistical data from the greenhouse horticulture industry
Villanueva-Rey et al. (2014)	Wine grape	Spain	1 organic (biodynamic)/1 conventional vineyard
Warner et al. (2010)	Strawberry	UK	Total of 20 farms comprising 3 organic/6 conventional strawberry production systems
Williams et al. (2006)	Milk, beef, pig, poultry, wheat, oilseed rape, potato, tomato	UK	Farm statistical data (official UK and private company data), literature, expert judgment, existing inventories including ecoinvent
Williams et al. (2010)	Wheat, potato	UK	National survey data/literature data
Zafiriou et al. (2012)	Asparagus	Greece	3 organic/5 conventional farms

production systems to investigate how different LCA modeling approaches can influence the results of milk carbon footprints (Flysjö et al., 2012).

3.1.2. Functional unit

Except for the studies of Kavargiris et al. (2009), Litskas et al. (2011), Michos et al. (2012) and Zafiriou et al. (2012) where impacts were related to area only, all of the reviewed studies expressed environmental impacts of the impact categories listed in Table 2 as impact per product unit. Three of the studies analyzing milk (Haas et al., 2001; van der Werf et al., 2009; Hörtenhuber et al., 2010), the study of Nemecek et al. (2011a) on arable crops, and the study of Abeliotis et al. (2013) on beans additionally expressed the environmental impacts by area and year.

3.1.3. Data basis and sample size

Of the 34 studies, 22 based their comparison on production data from a sample of real farms (Table 1). Those studies comparing production systems on nationwide scale used average national statistical data (Basset-Mens and van der Werf, 2005; Williams et al., 2006, 2010; Meisterling et al., 2009; Hörtenhuber et al., 2010; Leinonen et al., 2012a,b; Venkat, 2012). One study used statistical data from the horticultural industry (Vermeulen and van der Lans, 2011). One study on field crops used data from long term field trials (Nemecek et al., 2011a). In two cases regional production data was used (Juraske and Sanjuán, 2011; Venkat, 2012), one study compared products from model farms of which data origin was not further specified (Bos et al., 2007), and one study derived the average agricultural practice within a region from producers without mentioning their number (Abeliotis et al., 2013).

Overall, the data basis for production data regarding management practices, inputs, and yields in the reviewed comparative LCAs can be considered to be of high reliability. However, in 18 studies, data were taken from 10 or less farms for one or both farming systems (Table 1). In these cases it is questionable whether the results are representative for the farming system. In nine studies, the sample size of conventional farms was larger than the sample size of organic farms while sample size of organic farms was larger in three studies (Table 1). Nemecek et al. (2011a) compared arable crops from organic and conventional systems and calculated

Table 2
Overview of impact categories analyzed per product group and the relative differences between organic and conventional systems in the 26 reviewed studies.

Impact category	Relative difference organic/conventional on per area unit and year[a]	Relative difference organic/conventional on per product unit[a]	# of studies
Milk			11
Energy demand	−70 to −39%	−56 to −7%	8
Global warming potential (GWP)	−67 to −13%	−38 to +53%	10
Eutrophication potential	−76 to −2%	−66 to +63%	7
Acidification potential	−51 to −2%	−13 to +63%	7
Ecotox terrestrial	−73%	−59%	1
Pesticide use	−100 to −94%	−100 to −89%	3
Productivity	−47 to −6%		11
Land use		+6 to +90%	11
Beef			3
Energy demand	−64 to −22%	−35 to +53%	2
Abiotic resource use	−53%	−14%	1
GWP	−60 to −24%	−15 to +15%	3
Eutrophication potential (aquatic and terrestric combined)	+13%	+108%	1
Eutrophication potential terrestric	+12%	+42%	1
Eutrophication potential aquatic N	−8%	+17%	1
Eutrophication potential aquatic P	−26%	−6%	1
Acidification potential	−34 to +10%	+40 to +82%	2
Ozone vegetation	−61 to −22%	−1 to +8%	1
Ozone human	−58 to −21%	0 to +14%	1
Resource use K	−98 to −90%	−95 to −87%	1
Resource use P	−97 to −96%	−97 to −96%	1
Water use (blue water)	−59 to −33%	−15 to +14%	1
Productivity	−64 to −21%		3
Land use		+27 to +175%	3
Arable land use		−70 to −14%	1
Deforested land use		−98 to 0%	1
Pesticide use	−100%	−100%	1
Ecotox terrestrial incl. pesticides	−99 to −97%	−98 to −96%	1
Ecotox aquatic incl. pesticides	−100 to −99%	−99%	1
Human tox incl. pesticides	−95 to −74%	−86 to −67%	1
Pig			3
Energy demand	−50 to −23%	−13 to +40%	3
Abiotic resource use	−45%	−6%	1
GWP	−41 to −5%	−11 to +73%	3
Eutrophication potential (aquatic and terrestric combined)	−67 to −43%	−43 to +4%	2
Eutrophication potential terrestric	+24%	+116%	1
Eutrophication potential aquatic N	0%	+74%	1
Eutrophication potential aquatic P	−54%	−20%	1
Acidification potential	−81 to +12%	−67 to +96%	3
Ozone vegetation	−36%	+12%	1
Ozone human	−34%	+15%	1
Resource use K	−96%	−93%	1
Resource use P	−94%	−89%	1
Water use (blue water)	−45%	−4%	1
Productivity	−45 to −42%		3
Land use		+73 to +82%	3
Arable land use		+82%	1
Deforested land use		−97%	1
Pesticide use	−100 to −90%	−100 to −83%	2
Ecotox terrestrial incl. pesticides	−98%	−96%	1
Ecotox aquatic incl. pesticides	−99%	−98%	1
Human tox incl. pesticides	−92%	−98%	1
Poultry			4
Abiotic resource use	−60 to +56%	+80 to +241%	2
Energy demand	−64 to −32%	+3 to +59%	4
GWP	−71 to −33%	−24 to +46%	4
Eutrophication potential (aquatic and terrestric combined)	−46 to −20%	+76 to +140%	2
Eutrophication potential terrestric	+6%	+140%	1
Eutrophication potential aquatic N	−12%	+100%	1
Eutrophication potential aquatic P	−56%	0%	1
Acidification potential	−56 to −12%	+16 to +100%	4
Ozone vegetation	−48%	+18%	1
Ozone human	−56%	0%	1
Resource use K	−99%	−97%	1
Resource use P	−85%	−67%	1
Water use (blue water)	−93%	−85%	1
Productivity	−78 to −54%		4
Land use		+119 to +346%	4
Arable land use		+124%	1
Deforested land use		−83%	1

(*continued on next page*)

Table 2 (*continued*)

Impact category	Relative difference organic/conventional on per area unit and year[a]	Relative difference organic/conventional on per product unit[a]	# of studies
Pesticide use	−98 to −96%	−92 to −90%	2
Ecotox terrestrial incl. pesticides	−99%	−98%	1
Ecotox aquatic incl. pesticides	−100%	−99%	1
Human tox incl. pesticides	−93%	−83%	1
Eggs			1
Abiotic resource use	−47%	+122%	1
Energy demand	−63%	+56%	1
GWP	−72%	+17%	1
Eutrophication potential (aquatic and terrestric combined)	−52%	+104%	1
Acidification potential	−59%	+72%	1
Productivity	−76%		1
Land use		+323%	1
Pesticide use	−99%	−96%	1
Fruits & vegetables			13
Abiotic resource use	−89 to +42%	−71 to +89%	3
Energy demand	−48 to +54%	−25 to +104%	5
GWP	−90 to +121%	−81 to +130%	8
Eutrophication potential	−96 to +219%	−90 to +323%	3
Acidification potential	−94 to +127%	−83 to +201%	2
Ozone (photochemical oxidation)	−92 to −5%	−79 to +30%	2
Ozone depletion	−94 to −14%	−84 to +17%	2
Ecotox terrestrial	−100%	−99%	2
Productivity	−65 to +76%		12
Ecotox aquatic	−100%	−100%	1
Human tox	−100 to −82%	−100 to −76%	2
Nuts			1
GWP	+18 to +22%	+52 to +490%	1
Arable crops			8
Abiotic resource use	−77 to −17%	−83 to +22%	3
Energy demand	−77 to −21%	−56 to +14%	6
GWP	−69 to −92%	−41 to +45%	8
Eutrophication	−65 to +104%	−62 to +210%	5
Acidification	−84 to +119%	−58 to +66%	5
Ozone (photochemical oxidation)	−91 to −13%	−93 to +9%	2
Ozone depletion	+24 to +32%	0 to +11%	1
Resource use K	−75%	−66%	1
Resource use P	−97%	−96%	1
Pesticide use	−100 to −81%	−100 to −72%	2
Productivity	−68 to +32%		8
Land use		+9 to +214%	4
Ecotox terrestrial	−99 to +25%	−100 to +8%	2
Ecotox aquatic	−87 to −36%	−84 to −25%	1
Ecotox aquatic (freshwater)	−252 to +38%	−0.06 to +0.03%	1
Ecotox aquatic (marine)	+23 to +29%	−2 to +10%	1
Human tox	−65 to −17%	−50 to −2%	2

Environmental impacts on per area unit were calculated if not explicitly given in the studies.
[a] Basis: conventional.

the average yearly environmental impacts of different crop rotations with rotation cycles of 6 years. Villanueva-Rey et al. (2014) considered two years of production in their analysis of wine grapes. All other studies considered only one year of production.

3.1.4. Reported impacts

Three studies showed a higher productivity for organic production systems (Liu et al., 2010; Venkat, 2012; Abeliotis et al., 2013) and in one study the same productivity for organic and conventional was reported (Juraske and Sanjuán, 2011). Out of the 12 crops analyzed in Venkat (2012) higher productivity in organic was only reported for alfalfa, blueberry, raisin and wine grape and apple (for the latter two only in one out of two cases analyzed). In all other studies reviewed productivity of conventional production was higher.

Further, organic products usually had lower environmental impacts on a per area unit across all of the analyzed impact categories. The most noticeable exception was the study of Abeliotis et al. (2013) where impacts of organic beans were also higher on a per area basis for all impact categories analyzed except for aquatic ecotoxicity. The authors attributed the higher impacts to the higher diesel, water, and electricity input per ha in organic. Further exceptions were abiotic resource use, eutrophication and acidification potential for beef, pig and poultry production in Williams et al. (2006) and Alig et al. (2012), energy demand, eutrophication and acidification potential of tomatoes in Williams et al. (2006), eutrophication potential of wheat and potatoes in Williams et al. (2010), acidification potential of wheat and potatoes in Williams et al. (2010) and global warming potential of strawberries in Warner et al. (2010) (Table 2, see also Supplementary Material). For the same impact categories and the same commodity, the environmental impacts reported in the reviewed LCA studies varied considerably; e.g. the relative difference between the GWP of organically and conventionally produced milk was found to vary from −67 to −13% per area unit and from −38% to +53% per product unit (Table 2).

Table 3
Relative difference between environmental impacts per product unit of selected products from the ESU-services and ecoinvent v.2.2 databases.

	Relative difference organic/integrated on per product unit[a]			
Livestock products[b]	Milk	Beef	Pig	Poultry
Energy demand	−5%	−2%	−24%	−8%
Global warming potential (GWP)	−12%	−8%	−25%	−18%
Ozone depletion	−3%	−8%	−39	−17%
Eutrophication potential	−13%	−1%	+4%	+4%
Acidification potential	−12%	−13%	−30%	−21%
Heavy metals, water	−30%	−48%	−81%	−79%
Heavy metals, soil	−165%	−261%	+405%	−79%
Pesticide use	−100%	−99%	−100%	−100%
Water use	−69%	−76%	−73%	−73%
Land use	−1%	−23%	−32%	−32%
Fruits & vegetables[b]	Tomatoes	Carrots	Strawberries	Pear
Energy demand	−71%	+12%	+61%	+26%
Global warming potential (GWP)	−78%	−9%	+39%	+10%
Ozone depletion	−69%	−46%	+8%	−50%
Eutrophication potential	−17%	−69%	−65%	−85%
Acidification potential	−86%	+13%	+84%	+17%
Heavy metals, water	−97%	−60%	−25%	+60%
Heavy metals, soil	+306%	+2410%	+5981%	−29%
Pesticide use	−53%	−100%	−96%	−100%
Water use	−28%	+51%	+64%	+5%
Land use	+37%	−38%	−117%	−117%
Arable crops[c]	Barley grains	Soybeans	Wheat grains	Potatoes
Energy demand	−6%	−10%	−11%	−5%
Global warming potential (GWP)	+18%	−12%	−9%	+88%
Ozone depletion	−66%	−54%	−81%	−68%
Eutrophication potential	+54%	−26%	+80%	+39%
Acidification potential	−57%	−59%	−59%	−9%
Heavy metals, water	−77%	−65%	−79%	−54%
Heavy metals, soil	+333%	−105%	+665%	+1102%
Pesticide use	−100%	−100%	−100%	−100%
Water use	−65%	−54%	−68%	−12%
Land use	0%	−36%	−4%	+1%

[a] Basis: conventional.
[b] Inventories from LCI database of ESU-services only (Jungbluth et al., 2013).
[c] Inventories from ecoinvent v2.2 (Nemecek et al., 2007).

The relative differences between organic and integrated products from the ESU-services Ltd. (Jungbluth et al., 2013) and ecoinvent (v2.2) databases (Nemecek et al., 2007) are listed in Table 3. Impacts were calculated with the ecological scarcity method (Frischknecht et al., 2009; Jungbluth et al., 2012). The differences listed in Table 3 are within the ranges found in the comparative studies (Table 2) for the respective product and impact category or are slightly better for organic: with the exceptions of energy demand for pig and poultry; GWP for pig; eutrophication potential for beef and all fruits and vegetables; acidification potential of tomatoes; and land use of livestock products, fruit and vegetables but without tomatoes, soybean and wheat. Land use impacts for livestock products and fruit and vegetables are less for organically produced products because the biodiversity on the organic fields is higher, which is accounted for in the ecological scarcity method and thus balances the higher land occupation due to lower yields.

3.1.5. Interpretation of results

Regarding the interpretation of results, only six of the 34 reviewed studies conducted a sensitivity analysis on the choices of emission models or the choices of impact assessment methods, and only seven studies carried out a Monte-Carlo simulation to verify uncertainties within the results (Table 4). Six of the 34 reviewed studies concluded that organic farming systems compared to conventional perform better in some impact categories (e.g., non-renewable energy use, GWP, resource use of P and K, ecotoxicity) and worse in others (e.g., GWP, eutrophication and acidification potential) (Table 4). Eighteen studies concluded that organic farming has lower environmental impacts, or may have lower impacts in certain cases, for the impact categories analyzed. However, five of these 18 studies referred this conclusion to impacts per area only. Two studies concluded that there are no differences in environmental impacts at product level between organic and conventional farming systems. Finally, four studies drew no conclusions on the environmental performance of the analyzed farming systems because either the focus was on the assessment procedure or no generalization was possible due to small sample sizes.

3.2. Critical points within the goal and scope definition

Of the reviewed LCA studies, 31 were attributional and three claimed to have considered a consequential perspective (Liu et al., 2010; Kristensen et al., 2011; Flysjö et al., 2012). In attributional LCAs, the analysis gives a description of resource flows and emissions attributed to the functional unit assuming a status quo situation. Consequential LCAs follow a cause-effect chain approach to analyze how pollution and resource flows within a system change in response to change in the provision of the functional unit (Thomassen et al., 2008a). As Earles and Halog (2011) simply put it, consequential LCA represents the convergence of LCA and economic modeling methods. The choice of attributional or consequential LCA strongly determines the choice of co-product handling and, by that, the choice of system boundary. Physical relationships, exergy, energy, mass, or economic allocation are usually used in attributional LCA, while consequential LCA uses system expansion to determine the environmental burden to be attributed to co-products.

Flysjö et al. (2012) and Kristensen et al. (2011) used system expansion when determining the GWP of milk to distribute emissions between milk and meat. They argue that, when comparing organic with conventional milk production, it is important to consider the linkage between milk and beef production because the system specific difference leads to different functions (higher milk and lower meat production in the one case, lower milk and higher meat production in the other). These different functions are usually not considered by attributional LCA: In organic milk production systems cows on average have more lactation periods and therefore deliver more beef meat (Flysjö et al., 2012). Flysjö et al. (2012), in their Swedish study, calculated that 5 g more meat (carcass weight) were produced per kg of organically produced energy corrected milk (ECM). Assuming constant consumption patterns, these 5 g of extra meat per kg of milk have to be compensated by alternative conventional meat production systems: depending on the specific socio-economic context. For Sweden, Flysjö et al. (2012) assumed that beef from suckler cow systems would replace it. Kristensen et al. (2011) assumed, in the case of Denmark, that 50% would be replaced by pork and 50% by beef from suckler cows and intensive steer production.

In the context of GWP mitigation measures, increasing milk yield per cow is a solution to reduce emissions per unit of milk that is often discussed (see Flysjö et al. (2012) and the studies cited therein). This conclusion is mostly based on attributional LCAs that allocate GHG emissions between milk and beef and thereby ignore the link between milk and beef production. Interestingly, when considering the linkage between milk and beef production through system expansion with a consequential perspective, no correlation between GHG emissions per unit of milk and the milk annual yield per cow exists (Flysjö et al., 2012; Zehetmeier et al., 2012). In contrast to attributional LCAs, consequential LCAs have been suggested for the assessment of animal production systems because they can provide insight into the multidimensional, and sometimes

Table 4
Sensitivity and uncertainty analyses of results and main conclusions drawn in the reviewed studies.

Study	Sensitivity analysis on choices of methods/models	Uncertainty analyses of results	Main conclusions regarding farming systems
Abeliotis et al. (2013)	No	No	Integrated agricultural (IP) bean production is preferable among conventional, IP and organic in terms of acidification, eutrophication, and GWP. Organic bean production leads to the protection of abiotic resources.
Alig et al. (2012)	Yes	Yes	Compared to conventional meat production systems organic systems show a lower resource use of P and K and a lower terrestric and aquatic ecotoxicity due to the ban of mineral fertilizers and synthetic pesticides. However, lower yields in organic leads to higher environmental impact per kg meat.
Basset-Mens and van der Werf (2005)	No	Yes	No conclusion on farming systems (focus is on the scenario-based assessment procedure to compare different production systems).
de Backer et al. (2009)	No	No	Assessed on area basis organic farming shows a more favorable environmental profile than conventional farming. Due to lower yields in organic farming overall environmental benefits are strongly reduced or disappear on a per product basis.
Boggia et al. (2010)	No	No	System comparison showed that organic systems present the lowest environmental impacts.
Bos et al. (2007)	No	No	Organic dairy farming performs better and organic crop production worse than their conventional counterparts.
Casey and Holden (2006)	No	Yes	Shift from conventional to organic suckler-beef production would reduce GHG emissions in terms of product and area, but at the cost of a large drop in production per hectare.
Cederberg and Mattsson (2000)	No	Yes	Organic (i.e. extensive) milk production has environmental benefits (reduced use of pesticides and phosphorus). However, measures to reduce impacts in GWP, acidification and eutrophication have to be implemented for organic and conventional milk production.
Cederberg and Flysjö (2004)	n.a.[a]	No	Two strategies for reducing environmental impacts of milk production: 1) increasing production per cow while optimizing use of input resources (to be favored when land resources are limited). 2) extensive production, e.g. by organic farming (to be favored when land resources are sufficient for large home-based fodder production).
Flysjö et al. (2012)	Yes[b]	No	Increased milk production per cow does not necessarily reduce the GWP of milk when the alternative production of the by-product beef is considered.
Grönroos et al. (2006)	Yes[b]	No	Organic milk and rye bread production in Finland are somewhat less dependent on non-renewable energy sources than conventional. Changing from conventional to organic would be the easiest way to reduce non-renewable energy use in milk production. For rye bread it would be the second best choice since reduction potential within bakeries is even greater.
Guerci et al. (2013)	No	No	Huge variability in environmental impact within farms of a particular farming system due to different structural characteristics and management strategies. No upscaling of results on regional or national level possible due to small sample size. Proportion of grassland in the farming system and the feed efficiency in the herd most strongly influenced the environmental impact.
Haas et al. (2001)	No	No	LCA is suitable to compare farms and farming systems, but further development in methodology is needed.
Hörtenhuber et al. (2010)	No	No	Organic milk production systems have a lower GWP per ha of farmland and per kg of milk. However, site-specific conditions are important: The higher the potential milk output per cow, the lower the differences between compared systems.
Juraske and Sanjuán (2011)	No	No	Organic orange production represents the least toxic pest management alternative for human toxicity and fresh-water ecotoxicity impacts compared to integrated pest management (conventional production).
Kavargiris et al. (2009)	No	No	GWP (of fossil energy only) and non-renewable energy use in organic vineyards is lower than in conventional (on a per area basis). Organic farming systems could be an answer to the objectives of EEB's vision for European Agriculture (2008–2020).
Knudsen et al. (2010)	Yes	No	Organic soybeans imported from China to Denmark have lower environmental impact per ton produced than conventional soybeans. However, the transport stage accounts for 51% of GWP.
Kristensen et al. (2011)	No	No	There is a high variation in GWP per kg milk between farms within organic and conventional agriculture. Differences between the average GWP per kg milk from organic and conventional production was negligible.
Leinonen et al. (2012a)	No	Yes	Improving feed efficiency (quantity, composition, nutrient content) has the potential to reduce environmental impacts of broiler production.
Leinonen et al. (2012b)	No	Yes	Large differences in many impact categories between the different egg production systems analyzed. These reflect the differences in efficiency in production, feed consumption, and material and energy use. Further, there large variation in impacts between different production units within the same system can be observed.
Litskas et al. (2011)	No	No	Organic cherry production is an efficient way to reduce non-renewable energy input and GHG emissions (of fossil energy and fertilizer production only) in Natura 2000 sites (on a per area basis).
Liu et al. (2010)	Yes	No	Conversion from conventional to organic farming may contribute to the reduction of GHG emissions and non-renewable energy use.
Meisterling et al. (2009)	No	No	When conventional and organic wheat are transported the same distance to market, the organic wheat system produces less GHG emissions. Farming practices such as fuel use, fertilizer management, and tillage matter greatly when discussing the difference between organic and conventional products.
Michos et al. (2012)	No	No	Organic farming holds is an efficient way to reduce (on a per area basis) energy inputs and greenhouse gas-emissions (of fossil energy and fertilizer production only).
Nemecek et al. (2011a)	Yes	No	An overall assessment of organic crops in comparison to integrated crop production (conventional) led to the conclusion that environmental impacts of organic farming are in general equal or lower than impacts of conventional farming.

Table 4 (*continued*)

Study	Sensitivity analysis on choices of methods/models	Uncertainty analyses of results	Main conclusions regarding farming systems
Thomassen et al. (2008b)	No	No	Organic farms showed lower non-renewable energy use and lower eutrophication potential per kg of milk than conventional farms, but had higher GWP and acidification potential implying that higher NH_3, CH_4 and N_2O emissions occur on farm per kg of organic milk.
van der Werf et al. (2009)	No	Yes	Organic farms have lower potential environmental impacts than conventional farms per ha of land occupied, but there are no significant differences in impacts per kg of milk (except for land occupation).
Venkat (2012)	No[c]	No	Average emissions for organic production are higher by 10.6% due to lower yields, higher on farm energy use, the production and delivery of large quantities of compost and the fact that emissions from manufacture of synthetic fertilizers and pesticides used in conventional farming are not large enough of offset the additional emissions in organic farming.
Vermeulen and van der Lans (2011)	No	No	No conclusion on farming systems (study focused on the use of combined heat and power [cogeneration] within organic and conventional tomato production).
Villanueva-Rey et al. (2014)	No	No	Biodynamic viticulture shoed a substantially lower environmental profile for all assessed impacts (except for land use).
Warner et al. (2010)	No	No	It is possible to grow strawberries in low-input systems if cropped in season (without covers), if sufficient land is available to permit a long rotation and if suitable soil conditions are present.
Williams et al. (2006)	No	No	Organic field crops and animal products mostly consume less primary energy than the respective conventional products (except poultry meat and eggs). Regarding GWP, acidification, and eutrophication, organic production often results in increased burdens.
Williams et al. (2010)	No	No	Results for conventional production were similar to those from other European studies. However, values for organic systems were higher for the UK compared to other European studies.
Zafiriou et al. (2012)	No	No	Although organic farms showed a great variability regarding GWP (of fossil energy and fertilizer production only) and non-renewable energy use of asparagus production, organic farming can efficiently reduce energy inputs and GHG emissions.

[a] Life cycle inventory (LCI) only.
[b] By-product handling.
[c] However, sensitivity analysis on variable distance for transport of inputs to the farm was carried out.

conflicting, consequences of different mitigation options (de Boer et al., 2011). Especially when policy related questions with respect to sustainable food systems are addressed, consequential LCA may help to better understand the interrelations of the different farming systems with the market situation and consumption patterns, including the coverage of rebound effects, and by that may lead to more precise conclusions with regard to improvement strategies. Schader et al. (2012) pointed out that the consequential perspective seems to be important in particular in agricultural LCAs as it is better able to catch differences between farming systems: in particular when conclusions are generalized or used as a basis for decision-making by policy makers. This also applies in the context of comparisons between intensive with extensive farming systems.

Flysjö et al. (2012) also applied a consequential approach in the context of calculating GHG emissions from indirect land use change (ILUC). Two attributional approaches to calculating GHG emissions for soy meal production from Brazil were compared with two consequential approaches to calculate GHG emissions from land use: under the assumption that all land occupation is associated with GHG emissions. ILUC caused by displacement of crops to be grown in other countries is not considered in attributional approaches, whereas an evaluation of this is attempted in the consequential approaches. However, as argued in Flysjö et al. (2012), a limitation of the consequential approaches is that, in the case of Schmidt et al. (2011), the assessment of land use change (LUC) is based merely on land's biological production capacity, which is a great simplification. The method proposed in Audsley et al. (2009) is even more simplified since the same LUC-factor is used for all land. In the real world, decisions on land use and land use change are affected by many factors including economic market conditions, trade patterns and environmental regulations (Flysjö et al., 2012).

To identify options for reducing fossil energy use and GHG emissions, Liu et al. (2010) calculated GWP and energy use for organic and conventional pear production chains in two different regions in China. They used the consequential approach by Dalgaard and Halberg (2007) to distribute the environmental burden of farmyard manure between animal and plant production. Dalgaard and Halberg (2007) argue that the livestock products should be burdened with these extra emissions because manure in plant production causes higher N-emissions than mineral fertilizer if the yield level is set constant. However, to acknowledge the benefit of avoiding the production of mineral fertilizer by using farmyard manure, they also subtracted the emissions from the avoided production of mineral fertilizer from the burden of the livestock products. While citing this approach Liu et al. (2010) argue that, in their analyzed organic pear production chains, they do not need to account for the field emissions of farmyard manure because these burdened the livestock products. However, Dalgaard and Halberg (2007) only burdened livestock production with the extra N-emissions caused by the farmyard manure (compared to mineral fertilizer). Plant production still has to be burdened with the amount of N equivalent to the amount of N in the avoided mineral fertilizer. This is most probably the reason why, in the study of Liu et al. (2010), GWP of one ton of organic pears was much lower than for conventional pears: even though N input was higher in the organic pear production systems.

A comparison between different farming systems may become biased in cases where the allocation rule misses reflecting system-specific differences. To improve the quality of comparative LCAs for different agricultural systems, in particular if the aim is to answer policy related questions on what kind of agriculture to support, we suggest using system expansion whenever possible because agricultural production is often associated with co-production and a consequential approach might even better encompass the system under study. Furthermore, if the multi-functionality of agriculture is to be integrated in an assessment, the inclusion of non-commodity outputs is probably easier to accomplish by system expansion.

Table 5
Relation of N-surplus and calculated N-emissions in different studies of milk.

	Haas et al. (2001)	Cederberg and Mattsson (2000)	van der Werf et al. (2009)	Cederberg and Flysjö (2004)	Thomassen et al. (2008a,b)
N-input[a] organic [kg N/ha a^{-1}]	93	75	73	103	156
N-input conventional [kg N/ha a^{-1}]	128	235	152	224	288
Atmospheric deposition organic [kg N/ha a^{-1}]	20	10	0	8	30
Atmospheric deposition conventional [kg N/ha a^{-1}]	20	10	0	8	26
Total N-input organic [kg N/ha a^{-1}]	113	85	73	111	186
Total N-input conventional [kg N/ha a^{-1}]	148	245	152	232	314
Total N-output[b] organic [kg N/ha a^{-1}]	31	20	35	32	82
Total N-output conventional [kg N/ha a^{-1}]	48	47	64	66	91
N-use efficiency organic [output:input]	27	24	48	29	44
N-use efficiency conventional [output:input]	32	19	42	28	29
NH_3-N organic [kg N/ha a^{-1}]	55	24	13	25	28
NH_3-N conventional [kg N/ha a^{-1}]	68	61	16	39	40
NO_3-N organic [kg N/ha a^{-1}]	31	19	31	26	21
NO_3-N conventional [kg N/ha a^{-1}]	80	32	69	32	64
N_2O-N organic [kg N/ha a^{-1}]	4	1.2	3	3.2	5
N_2O-N conventional [kg N/ha a^{-1}]	6	3.1	4	4.7	7
NO-N organic [kg N/ha a^{-1}]	n.s.	n.s.	n.s.	n.s.	n.s.
NO-N conventional [kg N/ha a^{-1}]	n.s.	n.s.	n.s.	n.s.	n.s.
NO_x-N organic [kg N/ha a^{-1}]	7	n.s.	2	n.s.	n.s.
NO_x-N conventional [kg N/ha a^{-1}]	17	n.s.	3	n.s.	n.s.
Milk yield organic [kg/ha a^{-1}]	4882	3297	4416	5100	8937
Milk yield conventional [kg/ha a^{-1}]	7153	7415	7197	9460	14,713
N-surplus[c] organic [kg N/ha y^{-1}]	51	65	38	79	104
N-surplus conventional [kg N/ha y^{-1}]	100	198	88	166	223
Ratio surplus conventional: organic	1.96	3.05	2.30	2.10	2.15
kg N-surplus/kg milk organic	0.010	0.020	0.009	0.015	0.012
kg N-surplus/kg milk conventional	0.014	0.027	0.012	0.018	0.015
Ratio surplus conventional: organic	1.34	1.35	1.41	1.13	1.30
Total N from losses[d] organic [kg N/ha]	97	44	49	54	54
Total N from losses conventional [kg N/ha]	172	96	92	76	111
Ratio losses conventional: organic	1.77	2.17	1.89	1.40	2.05
kg N-losses/kg milk organic	0.020	0.013	0.011	0.011	0.006
kg N-losses/kg milk conventional	0.024	0.013	0.013	0.008	0.008
Ratio losses conventional: organic	1.21	0.97	1.16	0.75	1.24
Share of N-surplus found in N-losses organic	190%	68%	128%	69%	52%
Share of N-surplus found in N-losses conventional	172%	49%	105%	46%	50%
Relative difference in reported eutrophication potential between organic and conventional milk on a per amount of product basis	−66%	+9%	−30%	+35%	−36%

[a] N-inputs at farm gate as seeds, feed, straw, mineral fertilizer, imported manure, N-fixation, cattle.
[b] N-outputs at farm gate as products, exported manure.
[c] Balance between N input (from fertilizers, feed import, N-fixation, N-deposition) and N output (as animal and plant products).
[d] Sum of NH_3, NO_3, N_2O, NO, NO_x.

3.3. *Critical points within the inventory analysis*

3.3.1. *Nutrient balances vs. calculated N flows within studies on milk*

Nutrient losses from the nitrogen cycle are responsible for many environmental impacts of modern agriculture (Cederberg and Mattsson, 2000) and affect the eutrophication and acidification potential, GHG emissions, and biodiversity. N-emissions result from the N-surplus on farms. N-flows are different in organic and conventional agriculture because external N inputs on conventional farms are usually higher (by mineral fertilizer use and a higher share of concentrates in feed rations), which results in a higher N-input per hectare. As a consequence of the higher N-input per hectare, a higher N-surplus per hectare is often also found on conventional farms (Hansen et al., 2000; Dalgaard et al., 2002; de Boer, 2003; Knudsen et al., 2006). Surplus N is the nitrogen that is potentially lost to the environment through different N-emissions. Therefore, total N-losses by emissions cannot exceed N-surplus.

Among the 34 reviewed studies, Haas et al. (2001) and van der Werf et al. (2009) determined farm gate nutrient balances on the inventory level as the starting point for their emissions' calculations, while in Cederberg and Flysjö (2004) and Cederberg and Mattsson (2000), farm gate nutrient balances were used as a reference to the modeled emissions, and so provided an indirect indication for the emissions of nitrogen and phosphorus. Data from nutrient balances and calculated N-losses are summarized from five studies on milk from the 34 reviewed LCAs where the necessary data were provided (Table 5).

In all five studies listed in Table 5, N-surplus per hectare on organic farms was two to three times lower than on conventional farms. When dividing the N-surplus per hectare by the milk yield per hectare, the amount of surplus-N per kg milk was still lower for organic milk in all five studies. This result suggests that the overall N-losses due to emissions per kg milk should also be lower in organic systems. Calculated N-losses per hectare still were lower for organic production across all five studies. However, this changed in two cases when the N-losses per hectare were divided by the milk yield per hectare (Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004). Cederberg and Mattsson (2000) reported that N-losses per milk yield per hectare became equal for organic and conventional and Cederberg and Flysjö (2004) reported that N-losses per milk yield in organic production systems exceeded those of conventional systems (Table 5). In these two studies, the eutrophication potential was reported as 9 and 35% higher per kg of organic milk respectively: even though the N surplus per kg milk was lower in the organic systems. As a consequence of the lower N surplus in the organic systems, eutrophication potential per kg of organic milk should be lower too.

Interestingly, when calculating the share of N-losses from N-surplus, the calculated N-losses did not equal N-surplus in any of the studies: neither for organic nor for conventional production systems (Table 5). In two cases the calculated N-losses exceeded the amount of N-surplus for both organic and conventional systems (Haas et al., 2001; van der Werf et al., 2009). In all other cases, the calculated N-losses for organic systems made up 52–69% of the N-surplus whereas the calculated N-losses only amounted to 46–50% of the surplus-N for conventional milk production in the corresponding cases.

This analysis indicates that the models for calculating N-losses in LCAs need to be improved to better account for the N-surplus and, in particular, that the models have to be adapted to better reflect organic production systems. Cederberg and Flysjö (2004) stress that models used to calculate N-losses are probably not fully adapted for organic production systems. Therefore, in comparative LCAs of farming systems, it should be critically examined whether a higher eutrophication and acidification potential per product unit in organic farming systems is really due to lower yields: as is often argued. In cases where the N-surplus per product unit in extensive farming systems is lower than in intensive farming systems, the eutrophication potential per product unit should also be lower. In fact, N-surplus could be used as a cross reference to check for plausibility of calculated N-losses.

3.3.2. *Nutrient balances vs. calculated N flows within ecoinvent processes*

To analyze how calculated N-losses correspond with surplus-N from N-balances in ecoinvent inventories (v2.2), we transformed the inventories for the four crops listed in Table 6 from emissions per kg to emissions per ha by multiplying the emissions in the inventory by the yield of the respective crop. Thereby, for wheat and barley, we considered that the inventories per kg comprised an economic allocation step between grains and straw. The inventories for the four crops represent Swiss agricultural practice and are available for organic and integrated production (IP). Furthermore, we determined N-input and -output as well as N-losses per ha from the data given in the inventories (Table 6). We then calculated the N-balance by considering the total N content for organic fertilizers (slurry, solid manure). The N-surplus and, accordingly, N-losses were always higher in the organic crops than in IP systems except for the case of soybeans (Table 6). However, in all inventories, independent of the farming system, N-losses calculated by emission models exceeded N-surplus by a factor of 1.7–3.6 (Table 6) whereas N from nitrate emissions made up 74–90% of total N losses. The main reason for this imbalance is probably that the nitrate leaching model used within the inventories also includes nitrogen mineralization from soil organic matter (Richner et al., 2006; Nemecek et al., 2007; Nemecek and Schnetzer, 2011). The additional nitrogen from mineralization is not considered in the nitrate emission calculation in the N-balances in Table 6. However, while N mineralization is considered for the calculation of the nitrate leaching potential, this is not the case in the model for calculating N_2O emissions. Furthermore, as described in Richner et al. (2006), the nitrate leaching model does not consider losses from denitrification (N_2 and N_2O). This actually means that some of the N in the losses is counted twice. From the situation outlined above, again it becomes obvious that N-emission models used to calculate N-losses within inventories need to be improved and adjusted to the actual N-flows within different agricultural systems.

3.3.3. *Differentiation of dietary N-flows within livestock production systems*

Important differences between extensive and intensive livestock production systems are different dietary compositions that lead to different environmental impacts. In particular, dietary composition affects N-excretion and thereby influences N-emissions from manure (Klevenhusen et al., 2011). The level of excreted N strongly depends on the relationship between the amount of crude protein (CP) that is fed and the amount of dietary N built into milk and body mass (Külling et al., 2001). Ruminal N use efficiency is determined by the optimal ratio of degradable carbohydrates and the CP content.

As our review revealed, the relationship between the N-content in the diet and the N-content in the excrement is hardly ever considered in LCA inventories and might be an important reason why some of the surplus N remains unaccounted for in LCAs of milk (see Section 3.3.1) and beef (see later this section). From the reviewed studies of milk and beef production, only van der Werf et al. (2009) considered that a higher protein content in the feed ration leads to a higher nitrogen content within the excrement. Ryan et al. (2011), in their study of dairy production systems showed that, even though an increase of N-input in the diet leads to an increase in N-output in the form of production (i.e. milk), 40% of the extra N-input was lost to the environment. Regardless of the production system, more than 70% of the N consumed is excreted via urine as ammonia (Ryan et al., 2011; Orr et al., 2012). Higher ammonia emissions affect the GWP, as well as the eutrophication and acidification potential. Since, in conventional agriculture, feed rations have a higher average protein content (Alig et al., 2012), higher ammonia emissions from excrement should be attributed to conventional agriculture. Orr et al. (2012) reported that a 2.7 fold increase of N-content in the diet led to a 1.8 fold increase of N excreted in urine. Cederberg and Mattsson (2000) concluded that the calculation of N-losses by emission models, and especially ammonia emissions, seemed to be the most uncertain. In their study, 90% of the acidification potential was due to ammonia losses in organic and conventional systems and, within the eutrophication potential, ammonia accounted for approximately 50%. A better adaptation of ammonia emission models to different farming systems, including taking the diet-related N-flows into account, would, therefore, lead to more accurate estimates for acidification, eutrophication and GWP within comparative LCAs of animal products.

The degree to which dietary compositions may influence N-flows in different farming systems could be demonstrated using the information provided in Alig et al. (2012), where organic and conventional steer production systems in Switzerland were compared. Even though different dietary compositions between organic and conventional steer production were considered to account for different environmental impacts from feed production and for different formations of enteric CH_4 in the cattle, the influence of different dietary compositions on the N-excretion was neglected. Instead, the same annual N-excretion rate per beef production unit of 33 kg N y^{-1} was assumed for organic and conventional steer production systems (Table 7). Based on this assumption, and due to the longer time needed to gain the slaughter weight in the organic system, 1 kg of beef (LW) in the organic system produced 43%

Table 6
N-balance vs. N-surplus in ecoinvent inventories (v2.2) of arable crops at Swiss farms [kg N/ha* a^{-1}].

	Wheat grains		Barley grains		Soybeans		Potatoes	
	Organic	IP	Organic	IP	Organic	IP	Organic	IP
Slurry spreading[a], by vacuum tanker/CH U	86.1	5.6	69.2	11.7	6.7	16.2	18.4	22.5
Solid manure[b] loading and spreading, by hydraulic loader and spreader/CH U	35.5	0.5	28.5	8.8	13.3	11.0	65.1	70.6
Seeds (organic, at regional storehouse/CH U, IP at regional storehouse/CH U respectively)	4.0	3.6	2.1	1.6	7.2	6.6	5.8	5.8
Ammonium nitrate, as N, at regional storehouse/RER U		67.1		48.5		0.0		16.9
Urea, as N, at regional storehouse/RER U		23.6		17.0		0.0		5.9
Diammonium phosphate, as N, at regional storehouse/RER U		7.0		6.6		0.0		1.0
Calcium ammonium nitrate, as N, at regional storehouse/RER U		33.7		24.3		0.0		8.5
Ammonium sulfate, as N, at regional storehouse/RER U		5.1		3.7		0.0		1.3
N-fixation[c]					143.5	150.0		
N-Deposition[d]	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0
Total N-input (sum of all above N-inputs)	*151*	*171*	*125*	*147*	*196*	*209*	*114*	*158*
Yield main product	79.3	129.6	61.6	101.3	168.4	176.0	68.7	113.3
Straw	16.9	12.3	12.7	15.6				
Total N-output	*96*	*142*	*74*	*117*	*168*	*176*	*69*	*113*
N-surplus	*54*	*29*	*50*	*30*	*27*	*33*	*46*	*44*
NH_3-N	27.9	7.5	24.1	7.9	2.8	4.7	12.9	15.1
NO_x-N	0.5	0.6	0.5	0.4	0.7	0.7	0.3	0.4
N_2O-N	3.1	3.9	3.0	2.7	4.4	4.7	2.2	2.6
NO_3-N	90.4	74.9	91.7	96.1	43.9	45.2	66.0	59.1
Total N-losses	*122*	*87*	*119*	*107*	*52*	*55*	*81*	*77*
Share of N-surplus found in N-losses [%]	224	296	236	352	190	169	179	174
kg N-losses/kg yield (main product)	0.030	0.014	0.029	0.016	0.018	0.019	0.004	0.002
kg N-surplus/kg yield (main product)	0.013	0.005	0.012	0.004	0.010	0.011	0.002	0.001

[a] Slurry composition according to Nemecek et al. (2005); N-content (N_{tot}) according to Walther et al. (2001); dilution 1:1.5.
[b] Solid manure composition according to Nemecek et al. (2005); N-content (N_{tot}) according to Walther et al. (2001).
[c] Assumed as 150 kg N/ha for conventional, yield adjusted for organic.
[d] Assumed as 25 kg/ha/a.

higher N-excretions than 1 kg of beef (LW) in the conventional system (Table 7). However, from the differences in dietary composition between the organic and the conventional steer production system, it is hardly possible that this would result in the same annual N-excretion rate. We, therefore, calculated the amount of N excreted in the two systems, based on the dietary compositions given in Alig et al. (2012), by adding the crude protein (CP) content of the different ingredients and subtracting the amount of N that was built into biomass. The latter was determined by summing the rumen degradable protein (RDP) of each dietary component. CP and RPD values were taken from the Swiss database on animal feed (http://www.feed-alp.admin.ch/start.php). Despite the higher digestibility of the CP in concentrates, this led to an annual N-excretion rate per beef production unit of 44 kg N a^{-1} in the conventional system vs. 34 kg N a^{-1} in the organic system. Relating the differences in N-excretion rates to the amount of N excreted per kg of beef (LW) results in a calculated difference between the organic and the conventional system of 18% (Table 7), which is half of the difference reported in Alig et al. (2012).

Table 7
N excretion rate assumed in Alig et al. (2012) for Swiss steer production systems and recalculated from dietary N intake.

	Swiss beef production		Relative difference[a]
	Conventional	Organic	
Age of slaughter [Mt] as given in study	15	22	
End of life weight [kg LW] as given in study	525	538	
N excretion per kg live weight as given in study [g N/kg LW]	79	112	43%
N-uptake based on CP[b] intake from roughage [kg N/cattle]	35.3	78.0	
N-uptake based on CP intake from concentrates [kg N/cattle]	36.1	7.2	
Total N-uptake based on CP intake [kg N/cattle]	71.4	85.2	
Total N retention in body mass [kg N/cattle]	11.6	12.7	
Total N excreted [kg N/cattle]	59.8	72.5	
Annual N excretion rate per production unit [kg N/a]	44	34	
N excretion per kg live weight [g N/kg LW]	114	135	18%

[a] Basis conventional.
[b] Crude protein.

Assuming the same annual N-excretion rate per animal in both systems would mean that, in the extensive system due to a rearing phase that is one and a half times longer, the cattle would also eat one and a half times more protein as in the intensive system. If this was the case, then the end of life weight within the extensive system should be considerably higher than in the intensive system: despite the lower fodder use efficiency in the extensive system (which is 0.16 vs. 0.2 in the intensive system) due to the higher share of roughage in the diet. By simplifying assumptions, a system difference was generated that was twice the difference calculated using N-excretion rates specific for organic and conventional farming.

As a consequence of different N-excretion rates between different farming systems, average *N*-values in manure can be expected to differ between organic and conventional agriculture as well. This in turn leads to different N-emissions from manure storage and from crop production. However, N contents in manure from organic and conventional agriculture were not differentiated in any of the comparative LCA studies reviewed.

In contrast to the observation that the diet related effects on N-flows have hardly been considered in LCAs so far, the influence of different diets on CH_4 production from enteric fermentation is often differentiated between organic and conventional milk and beef production systems (Table 8). Higher CH_4 emissions are attributed to organic agriculture due to forage based diets. However, if different CH_4 emissions are considered from enteric fermentation based on different diets, different CH_4 emissions

Table 8
Diet-related differentiation of enteric fermentation in LCAs on milk and beef.

	Total # of studies	# of studies differentiating emission factors for enteric fermentation	Emission factors based on		
			Kirchgeßner et al. (1993, 1995)	IPCC (2006) (tier 2)	Others
# of studies for milk	9	7	4	2	3
# of studies for beef	3	2	1	1	1

during manure storage should be considered as well. Concentrates in the diet increase the content of undigested nutrients in manure, which may be transformed to CH_4 by microbial degradation. This may compensate for diet-related mitigation achievements in the animal (Klevenhusen et al., 2011). Hindrichsen et al. (2006) showed that CH_4 emissions from slurry increased when dairy cows were fed mixed forage-concentrate diets instead of forage-only diets. Despite this evidence from experimental studies none of the reviewed comparative LCAs on milk and beef considered diet-dependent CH_4 emissions from manure during storage.

In contrast, newer studies even challenge the widespread assumption that forage-only diets necessarily result in higher enteric CH_4 formation than mixed forage-concentrate diets (Klevenhusen et al., 2011). This means that the GWP of forage-based milk and beef production systems in LCAs have been overestimated in those cases where different emission factors for enteric fermentation based on diet composition were used and where emissions of manure storage was not differentiated for diet compositions.

3.4. Critical points within the impact assessment

Of the 34 studies, 10 analyzed the global warming potential (GWP) only (carbon footprint (CF) studies) (Casey and Holden, 2006; Bos et al., 2007; Meisterling et al., 2009; Hörtenhuber et al., 2010; Liu et al., 2010; Warner et al., 2010; Kristensen et al., 2011; Vermeulen and van der Lans, 2011; Flysjö et al., 2012; Venkat, 2012) [see Supplementary Material for a tabular overview]. Five studies focused on energy demand (Grönroos et al., 2006; Kavargiris et al., 2009; Litskas et al., 2011; Michos et al., 2012; Zafiriou et al., 2012). In addition to energy demand Kavargiris et al. (2009), Litskas et al. (2011), Michos et al. (2012), and Zafiriou et al. (2012) also quantified greenhouse gas emissions from fossil fuel use and fertilizer production. However, N_2O emissions from soils were not included. From these studies we, therefore, only considered energy demand in this review. Further, one study examined toxicity (human toxicity and freshwater ecotoxicity) (Juraske and Sanjuán, 2011). The remaining 18 studies analyzed at least eutrophication and acidification potential in addition to GWP. Of these 18 studies, nine studies assessed a wider range of environmental impacts that can be routinely assessed today using LCA (Williams et al., 2006, 2010; de Backer et al., 2009; Nemecek et al., 2011a; Alig et al., 2012; Leinonen et al., 2012a,b; Abeliotis et al., 2013; Villanueva-Rey et al., 2014). Biodiversity impacts were assessed in Alig et al. (2012) and Nemecek et al. (2011a), using the LCIA-method “SALCA-BD” (Jeanneret et al., 2009, 2014), in Guerci et al. (2013) using biodiversity damage scores as proposed by De Schryver et al. (2010), and in Haas et al. (2001) where impact on biodiversity was qualitatively judged based on self-defined criteria. However, all four studies the impact on biodiversity was assessed for only part of the life cycle of the specific products and was determined on a per area unit only except in Guerci et al. (2013) where land use impacts were related to the production of 1 kg of milk. Impacts of cultivation practices on soil quality were assessed in Nemecek et al. (2011a) who applied the LCIA-method “SALCA-SQ” (Oberholzer et al., 2006) with impacts related to area.

The above analysis shows that comparative LCAs of agricultural products are far from a comprehensive environmental assessment. The environmental assessment was restricted to only one single impact category in almost half of the studies reviewed. However, important environmental impacts of farming systems, such as effects on biodiversity and soil quality, are not routinely assessed by LCA due to a lack of appropriate impact assessment methods and are, therefore, usually lacking in contemporary comparative LCAs (Cederberg and Mattsson, 2000; Reap et al., 2008; Finnveden et al., 2009; Schader et al., 2012). The impact on biodiversity was considered in only four studies (Haas et al., 2001; Nemecek et al., 2011a; Alig et al., 2012; Guerci et al., 2013). However, the applied impact assessment methods do not allow for a comprehensive assessment that covers the entire life cycle because they only assess the biodiversity impacts of the agricultural production phase.

A difference between organic and conventional farming is that no synthetic pesticides are used within organic farming. Toxicity related impacts (human toxicity, terrestrial ecotoxicity, freshwater and marine aquatic ecotoxicity) were reported in nine of the reviewed studies (Basset-Mens and van der Werf, 2005; de Backer et al., 2009; van der Werf et al., 2009; Boggia et al., 2010; Juraske and Sanjuán, 2011; Nemecek et al., 2011a; Alig et al., 2012; Abeliotis et al., 2013; Villanueva-Rey et al., 2014). Four different LCIA methods (CML, 2000, Eco-indicator 99, EDIP97, and USEtox) were used, including mid- and endpoint characterization methods. Impacts calculated for organic systems were always lower (−20% to −100%) than those reported for conventional systems except for the assessment of beans in Abeliotis et al. (2013) were higher impacts for terrestrial and aquatic (freshwater and marine) ecotoxicity were attributed to organic. The usually lower toxicity impacts in organic can mainly be explained by the usual application of synthetic pesticides in conventional systems which lead to higher toxicity scores. However, the availability of characterization factors for biological/natural and inorganic pesticides, which are partly registered for use in organic agriculture, is still generally lacking. Therefore, a thorough comparison of the two agricultural systems is not always possible and might underestimate the impacts in the organic system when these compounds are not included in the LCA: “lack of data” is the most stated reason in the reviewed studies. Recent developments, such as PestLCI 2.0 (Dijkman et al., 2012) on an inventory level and dynamiCROP (Fantke et al., 2011) on the impact assessment level, in combination with an increasing availability of physicochemical and toxicological data, might help in improving the analysis of toxic impacts due to emission of pesticides in the future.

4. Conclusions

LCA, by definition, does not compare products but product systems (ISO/DIS, 2006). As for a certain industrial product an agricultural product, too may be produced in different production processes, i.e. different farming systems. In this sense, a comparison of organic vs. conventional products is inevitably a comparison of organic vs. conventional farming systems. So, the comparison of the environmental impact of organic vs. conventional products using

LCA must reflect the impacts of these different ways of production adequately.

However, from the 34 reviewed LCA studies, which compared products from organic and conventional farming systems, it is not yet possible to draw a conclusive picture on the general environmental performance of the different farming systems. An important reason for this is that comparative LCAs on agricultural products from different farming systems often do not adequately differentiate the specific characteristics of organic and conventional farming on the inventory level. This is in accordance with the conclusion from an expert workshop on the "Definition of Best Indicators for Biodiversity and Soil Quality for Life Cycle Assessment (LCA)", which pointed out the importance of more detailed assessments to illustrate the effects of different management practices (e.g. organic vs. conventional crops) (Milà i Canals et al., 2006). For example, the nitrogen emissions' calculations are especially often based on the same assumptions for both farming systems although different assumptions should actually be taken. Often assumptions taken for the organic system are based on the values for conventional agriculture. Unfortunately the adaptation of emission models to extensive farming systems is sometimes hindered by a lack of reliable background data from these systems.

Regarding the assessment and comparison of products from different farming systems with LCA we identified potential for methodological improvements at two levels:

First, physical relationships between agricultural products and environmental effects as accounted for in attributional LCAs need to be differentiated more precisely between farming systems and more comprehensively regarding the relevant impact categories. Certainly, there are differences between farming systems that can easily be incorporated in LCA as for example different inputs (fertilizers, pesticides, etc.) used within the two farming systems. Some differences between organic and conventional farming systems, though, are still rather difficult to be integrated in LCA as for example effects on biodiversity and soil quality or the multifunctionality of agriculture. Agriculture is widely seen as a multifunctional production process (OECD, 2001) which, in addition to food, feed and resources for energy production, provides noncommodity outputs such as landscape provision and ecosystem services to society. However, contemporary LCA studies focus on the environmental friendliness of agricultural products: expressing impacts per unit of product without allocation between commodity and non-commodity outputs. This narrow view, which focuses mainly on production efficiency, may often favor products from intensive production systems, although these systems have been shown by other assessment methods to be not environmentally sustainable (Gibbs et al., 2009; Geiger et al., 2010; Meehan et al., 2011). For an LCA-based comparison of farming systems beyond product level, it is necessary to either use different functional units to acknowledge these multifunctional outputs or to allocate the environmental impacts to the whole set of outputs that agriculture provides (Schader et al., 2012).

Second, to answer questions on environmental impacts of agricultural production systems that go beyond the physical relationships, i.e. policy- and environmental management related questions, consequential LCA approaches need to be considered that incorporate economic phenomena such as market elasticity, rebound effects, etc. Even though, it is not always straightforward to integrate economic phenomena into an engineering approach such as LCA.

Conclusions that have been drawn on the environmental performance of organic and conventional farming systems, based on comparative LCAs, should be reconsidered in light of the shortcomings identified within this review. Future comparative LCAs of farming systems must be improved accordingly.

5. Recommendations

Based on our analyses within this review, we suggest the following recommendations to improve LCA for comparison of products from different farming systems:

1. Since N-fluxes may be different between farming systems (in particular between extensive and intensive systems) and the human induced nitrogen cycle has environmental impacts on different levels, N-fluxes should be differentiated in more detail. Using N-balances at farm, farming branch, or field level could serve as a cross-reference for calculated N-losses.
2. In animal production systems, different production intensities are reflected in the feed ration composition, which leads to different nutrient and C-composition within the excrement. Thus, emissions during farmyard manure management and farmyard manure fertilization in plant production should be adjusted to the respective production intensity: as is often done for enteric fermentation.
3. There is a need to improve N- and C-emission models for farmyard manure management and fertilization. Changes in N- and C-stocks in soils are influenced by different farming practices and fertilizer types, which should be reflected in the emission models. First suggestions have been made for the modeling of N_2O-emissions from soils (Meier et al., 2012, 2014).
4. More background data on extensive farming systems should be generated and compiled so as to be available for use within LCA inventories (e.g. representative concentrations of nitrogen within farmyard manure from organic farms, nitrogen content within organic plant products, nitrogen excretion rates of animals under different feeding intensities, reliable CH_4 and N_2O emission measurements from farmyard manure storage under different feeding regimes [concentrate vs. forage based rations]). In cases where no reliable background data is available, and data from intensive farming systems are taken instead; this should be clearly stated.
5. Consequential LCA approaches should be used in cases where LCA is used for analyzing different agricultural production systems to find answers for policy-making or strategic environmental planning. Accordingly, system expansion should be applied for co-product handling to fully account of the different functions of the analyzed farming systems.

Acknowledgments

We thank the Coop Sustainability Fund, Basel, Switzerland for funding this study. Further, we are indebted to Florian Leiber (Research Institute of Organic Agriculture, Frick, Switzerland) for his assistance on animal nutrition, to Frank Hayer (Federal Office for the Environment, Bern, Switzerland), and Vanessa Geier (Research Institute of Organic Agriculture, Frick, Switzerland) for their critical comments on the manuscript and to Thomas Nemecek and Martina Alig (Agroscope, Zurich, Switzerland) for critically commenting those parts of the manuscript referring to their work. Finally, we would also like to thank two independent reviewers for their constructive comments and suggestions on the manuscript.

Appendix A. Supplementary material

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2014.10.006.

References

Abeliotis, K., Detsis, V., Pappia, C., 2013. Life cycle assessment of bean production in the Prespa National Park, Greece. J. Clean. Prod. 41, 89–96.

Alig, M., Grandl, F., Mieleitner, J., Nemecek, T., Gaillard, G., 2012. Ökobilanz von Rind-, Schweine- und Geflügelfleisch. Forschungsanstalt Agroscope Reckenholz-Tänikon ART, Zürich, p. 151.
Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, D., Webster, C., Williams, A., 2009. How Low Can We Go? An Assessment of Greenhouse Gas Emissions from the UK Food System and the Scope to Reduce Them by 2050. WWF-UK.
Basset-Mens, C., van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agric. Ecosyst. Environ. 105, 127–144.
Boggia, A., Paolotti, L., Castellini, C., 2010. Environmental impact evaluation of conventional, organic and organic-plus poultry production systems using life cycle assessment. World's Poultry Sci. J. 66.
Bommarco, R., Kleijn, D., Potts, S.G., 2013. Ecological intensification: harnessing ecosystem services for food security. Trends Ecol. Evol. 28, 230–238.
Bos, J.F.F.P., de Haan, J.J., Sukkel, W., Schils, R.L.M., 2007. Comparing energy use and greenhouse gas emissions in organic and conventional farming systems in the Netherlands. In: 3rd QLIF Congress, Hohenheim, Germany, pp. 1–4.
Casey, J.W., Holden, N.M., 2006. Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. J. Environ. Qual. 35, 231–239.
Cederberg, C., Flysjö, A., 2004. Life cycle inventory of 23 dairy farms in south-western Sweden. Swed. Inst. Food Biotechnol. 59.
Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production – a comparison of conventional and organic farming. J. Clean. Prod. 8, 49–60.
Dalgaard, R., Halberg, N., 2007. How to account for emissions from manure? Who bears the burden?. In: Proceedings from the 5th International Conference 'LCA in Foods', Gothenburg, Sweden.
Dalgaard, T., Heidmann, T., Mogensen, L., 2002. Potential N-losses in three scenarios for conversion to organic farming in a local area of Denmark. Eur. J. Agron. 16, 207–217.
de Backer, E., Aertsens, J., Vergucht, S., Steurbaut, W., 2009. Assessing the ecological soundness of organic and conventional agriculture by means of life cycle assessment (LCA). A case study of leek production. Br. Food J. 111, 1028–1061.
de Boer, I.J.M., 2003. Environmental impact assessment of conventional and organic milk production. Livest. Prod. Sci. 80, 69–77.
de Boer, I.J.M., Cederberg, C., Eady, S., Gollnow, S., Kristensen, T., Macleod, M., Meul, M., Nemecek, T., Phong, L.T., Thoma, G., van der Werf, H.M.G., Williams, A.G., Zonderland-Thomassen, M.A., 2011. Greenhouse gas mitigation in animal production: towards an integrated life cycle sustainability assessment. Curr. Opin. Environ. Sustain. 3, 1–9.
de Ponti, T., Rijk, B., van Ittersum, M.K., 2012. The crop yield gap between organic and conventional agriculture. Agric. Syst. 108, 1–9.
De Schryver, A.M., Goedkoop, M.J., Leuven, R.S.E.W., Huijbregts, M.A.J., 2010. Uncertainties in the application of the species area relationship for characterisation factors of land occupation in life cycle assessment. Int. J. Life Cycle Assess. 15, 682–691.
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, 973–986.
Earles, J.M., Halog, A., 2011. Consequential life cycle assessment: a review. Int. J. Life Cycle Assess. 16, 445–453.
Fantke, P., Juraske, R., Antón, A., Friedrich, R., Jolliet, O., 2011. Dynamic multicrop model to characterize impacts of pesticides in food. Environ. Sci. Technol. 45, 8842–8849.
Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. J. Environ. Manag. 91, 1–21.
Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2012. The interaction between milk and beef production and emissions from land use change – critical considerations in life cycle assessment and carbon footprint studies of milk. J. Clean. Prod. 28, 134–142.
Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–574.
Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342.
Frischknecht, R., Steiner, R., Jungbluth, N., 2009. The Ecological Scarcity Method – Eco-factors 2006: a Method for Impact Assessment in LCA. Federal Office for the Environment (FOEN), Zürich und Bern.
Geiger, F., Bengtsson, J., Berendse, F., Weisser, W.W., Emmerson, M., Morales, M.B., Ceryngier, P., Liira, J., Tscharntke, T., Winqvist, C., Eggers, S., Bommarco, R., Pärt, T., Bretagnolle, V., Plantegenest, M., Clement, L.W., Dennis, C., Palmer, C., Oñate, J.J., Guerrero, I., Hawro, V., Aavik, T., Thies, C., Flohre, A., Hänke, S., Fischer, C., Goedhart, P.W., Inchausti, P., 2010. Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. Basic Appl. Ecol. 11, 97–105.
Gibbs, K.E., Mackey, R.L., Currie, D.J., 2009. Human land use, agriculture, pesticides and losses of imperiled species. Divers. Distrib. 15, 242–253.
Grönroos, J., Seppälä, J., Voutilainen, P., Seuri, P., Koikkalainen, K., 2006. Energy use in conventional and organic milk and rye bread production in Finland. Agric. Ecosyst. Environ. 117, 109–118.
Guerci, M., Knudsen, M.T., Bava, L., Zucali, M., Schönbach, P., Kristensen, T., 2013. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. J. Clean. Prod. 54, 133–141.
Haas, G., Wetterich, F., Köpke, U., 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. Agric. Ecosyst. Environ. 83, 43–53.
Hansen, B., Kristensen, E.S., Grant, R., Høgh-Jensen, H., Simmelsgaard, S.E., Olesen, J.E., 2000. Nitrogen leaching from conventional versus organic farming systems – a systems modelling approach. Eur. J. Agron. 13, 65–82.
Hindrichsen, I.K., Wettstein, H.R., Machmüller, A., Kreuzer, M., 2006. Methane emission, nutrient degradation and nitrogen turnover in dairy cows and their slurry at different milk production scenarios with and without concentrate supplementation. Agric. Ecosyst. Environ. 113, 150–161.
Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems—model calculations considering the effects of land use change. Renew. Agric. Food Syst. 1–14.
IPCC, 2006. 2006 IPCC Guidelines for national greenhouse gas inventories. Agriculture, forestry and other land use Volume 4. Hayama, Japan.
ISO/DIS, 2006. ISO 14040:2006 Environmental Management – Life Cycle Assessment – Principles and Framework. Geneva, Switzerland.
Jeanneret, P., Baumgartner, D.U., Freiermuth Knuchel, R., Koch, B., Gaillard, G., 2014. An expert system for integrating biodiversity into agricultural life-cycle assessment. Ecol. Indic. 46, 224–231.
Jeanneret, P., Baumgartner, D.U., Freiermuth Knuchel, R., Gaillard, G., 2009. Methode zur Beurteilung der Wirkung landwirtschaftlicher Aktivitäten auf die Biodiversität für Ökobilanzen (SALCA-Biodiversität). Forschungsanstalt Agroscope Reckenholz-Tänikon ART, Zürich, p. 74.
Jungbluth, N., Itten, R., Stucki, M., 2012. Umweltbelastungen des Privaten Konsums und Reduktionspotenziale. ESU-services Ltd. im Auftrag des BAFU, Uster, CH.
Jungbluth, N., Flury, K., Doublet, G., Leuenberger, M., Steiner, R., Büsser, S., Stucki, M., Schori, S., Itten, R., 2013. Life Cycle Inventory Database on Demand: EcoSpold LCI Database of ESU-services. ESU-services Ltd., Zürich, Switzerland.
Juraske, R., Sanjuán, N., 2011. Life cycle toxicity assessment of pesticides used in integrated and organic production of oranges in the Comunidad Valenciana, Spain. Chemosphere 82, 956–962.
Kavargiris, S.E., Mamolos, A.P., Tsatsarelis, C.A., Nikolaidou, A.E., Kalburtji, K.L., 2009. Energy resources' utilization in organic and conventional vineyards: energy flow, greenhouse gas emissions and biofuel production. Biomass Bioenergy 33, 1239–1250.
Kirchgeßner, M., Roth, F.X., Windisch, W., 1993. Verminderung der Stickstoff- und Methanausscheidung von Schwein und Rind durch die Fütterung. Tierernährung 21, 889–120.
Kirchgeßner, M., Windisch, W., Müller, H.L., 1995. Nutritional factors for the quantification of methane production. In: Engelhardt, W.V., Leonhard-Marek, S., Breves, G., Giesecke, D. (Eds.), Ruminant Physiology: Digestion, Metabolism, Growth and Reproduction, Proceedings of the VIII International Symposium on Ruminant Physiology, pp. 333–348.
Klevenhusen, F., Kreuzer, M., Soliva, C.R., 2011. Enteric and manure-derived methane and nitrogen emissions as well as metabolic energy losses in cows fed balanced diets based on maize, barley or grass hay. Animal 5, 450–461.
Knudsen, M.T., Yu-Hui, Q., Yan, L., Halberg, N., 2010. Environmental assessment of organic soybean (Glycine max.) imported from China to Denmark: a case study. J. Clean. Prod. 18, 1431–1439.
Knudsen, M.T., Kristensen, I.S., Berntsen, J., Petersen, B.M., Kristensen, E.S., 2006. Estimated N leaching losses for organic and conventional farming in Denmark. J. Agric. Sci. 144, 135–149.
Kristensen, T., Mogensen, L., Knudsen, M.T., Hermansen, J.E., 2011. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livest. Sci. 140, 136–148.
Külling, D.R., Menzi, H., Kröber, T.F., Neftel, A., Sutter, F., Lischer, P., Kreuzer, M., 2001. Emissions of ammonia, nitrous oxide and methane from different types of dairy manure during storage as affected by dietary protein content. J. Agric. Sci. 137, 235–250.
Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012a. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: broiler production systems. Poult. Sci. 91, 8–25.
Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012b. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: egg production systems. Poult. Sci. 91, 26–40.
Litskas, V.D., Mamolos, A.P., Kalburtji, K.L., Tsatsarelis, C.A., Kiose-Kampasakali, E., 2011. Energy flow and greenhouse gas emissions in organic and conventional sweet cherry orchards located in or close to Natura 2000 sites. Biomass Bioenergy 35, 1302–1310.
Liu, Y., Langer, V., Høgh-Jensen, H., Egelyng, H., 2010. Life cycle assessment of fossil energy use and greenhouse gas emissions in Chinese pear production. J. Clean. Prod. 18, 1423–1430.
Meehan, T.D., Werling, B.P., Landis, D.A., Grattona, C., 2011. Agricultural landscape simplification and insecticide use in the Midwestern United States. Proc. Natl. Acad. Sci. U. S. A. 1–6.
Meier, M.S., Jungbluth, N., Stoessel, F., Schader, C., Stolze, M., 2014. Higher accuracy in N modeling makes a difference. In: 9th International Conference LCA of Food 8-10 October 2014. San Francisco, USA.
Meier, M.S., Schader, C., Berner, A., Gattinger, A., 2012. Modelling N_2O emissions from organic fertilisers for LCA inventories. In: Corson, M.S., van der

Werf, H.M.G. (Eds.), 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2012), 1–4 October 2012. INRA, Rennes, France, Sain-Malo, France, pp. 177–182.
Meisterling, K., Samaras, C., Schweizer, V., 2009. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. J. Clean. Prod. 17, 222–230.
Michos, M.C., Mamolos, A.P., Menexes, G.C., Tsatsarelis, C.A., Tsirakogloud, V.M., Kalburtji, K.L., 2012. Energy inputs, outputs and greenhouse gas emissions in organic, integrated and conventional peach orchards. Ecol. Indic. 13, 22–28.
Milà i Canals, L., Basson, L., Clift, R., Müller-Wenk, R., Bauer, C., Hansen, Y., Brandão, M. (Eds.), 2006. Expert Workshop on Definition of Best Indicators for Biodiversity and Soil Quality for Life Cycle Assessment (LCA). Proceedings and Conclusions. Centre for Environmental Strategy, University of Surrey, Guildford, UK.
Nemecek, T., Schnetzer, J., 2011. Methods of Assessment of Direct Field Emissions for LCIs of Agricultural Production Systems. Data v3.0 (2012). Agroscope Reckenholz-Tänikon Research Station ART, Zurich, Switzerland.
Nemecek, T., Heil, A., Huguenin, O., Meier, S., Erzinger, S., Blaser, S., Dux, D., Zimmermann, A., 2007. Life Cycle Inventories of Agricultural Production Systems. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.
Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011a. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104, 217–232.
Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., Schaller, B., Chervet, A., 2011b. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agric. Syst. 104, 233–245.
Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., 2005. Ökobilanzierung von Anbausystemen im Schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zurich.
Oberholzer, H.-R., Weisskopf, P., Gaillard, G., Weiss, F., Freiermuth, R., 2006. Methode zur Beurteilung der Wirkungen landwirtschaftlicher Bewirtschaftung auf die Bodenqualität in Ökobilanzen – SALCA-SQ. Agroscope FAL Reckenholz, Zurich, p. 98.
OECD, 2001. Multifunctionality. Towards an Analytical Framework. Organisation for Economic Co-operation and Development (OECD), Paris, p. 26.
Orr, R.J., Griffith, B.A., Cook, J.E., Champion, R.A., 2012. Ingestion and excretion of nitrogen and phosphorus by beef cattle under contrasting grazing intensities. Grass Forage Sci. 67, 111–118.
Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. Int. J. Life Cycle Assess. 13, 374–388.
Richner, W., Oberholzer, H.-R., Freiermuth, R., Huguenin, O., Walther, U., 2006. Modell zur Beurteilung des Nitratauswaschungspotentials in Ökobilanzen – SALCA Nitrate. Agroscope Reckenholz Tänikon (ART), Zürich.
Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461, 472–475.
Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90, 1–10.
Ryan, W., Hennessy, D., Murphy, J.J., Boland, T.M., Shalloo, L., 2011. A model of nitrogen efficiency in contrasting grass-based dairy systems. J. Dairy Sci. 94, 1032–1044.
Schader, C., Stolze, M., Gattinger, A., 2012. Environmental performance of organic farming. In: Boye, J.I., Arcand, Y. (Eds.), Green Technologies in Food Production and Processing. Springer, New York, pp. 183–210.
Schmidt, J., Reinhard, J., Weidema, B., 2011. Modelling of Indirect Land Use Change in LCA. Report v2. 2.-0. LCA Consultants, Aalborg.
Seufert, V., Ramankutty, N., Foley, J.A., 2012b. The yield performance of organic agriculture. In: Corson, M.S., van der Werf, H.M.G. (Eds.), 8th International Conference on Life Cycle Assessment in the Agri-food Sector (LCA Food 2012), 1–4 October 2012. INRA, Rennes, France, Sain-Malo, France, pp. 31–36.
Seufert, V., Ramankutty, N., Foely, J.A., 2012a. Comparing the yields of organic and conventional agriculture. Nature 485, 229–232.
Thomassen, M.A., Dalgaard, R., Heijungs, R., de Boer, I., 2008a. Attributional and consequential LCA of milk production. Int. J. Life Cycle Assess. 13, 339–349.
Thomassen, M.A., van Calker, K.J., Smits, M.C.J., Iepema, G.L., de Boer, I.J.M., 2008b. Life cycle assessment of conventional and organic milk production in the Netherlands. Agric. Syst. 96, 95–107.
Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? – A meta-analysis of European research. J. Environ. Manag. 112, 309–320.
van der Werf, H.M.G., Kanyarushoki, C., Corson, M.S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. J. Environ. Manag. 90, 3643–3652.
Venkat, K., 2012. Comparison of twelve organic and conventional farming systems: a life cycle greenhouse gas emissions perspective. J. Sustain. Agric. 36, 620–649.
Vermeulen, P.C.M., van der Lans, C.J.M., 2011. Combined heat and power (CHP) as a possible method for reduction of the CO_2 footprint of organic greenhouse horticulture. In: Dorais, M., Bishop, S.D. (Eds.), First International Conference on Organic Greenhouse Horticulture. Bleiswijk, Netherlands, pp. 61–68.
Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. J. Clean. Prod. 65, 330–341.
Walther, U., Ryser, J.P., Flisch, R., 2001. Grundlagen für die Düngung im Acker- und Futterbau. Agrarforschung 8, 1–80.
Warner, D.J., Davies, M., Hipps, N., Osborne, N., Tzilivakis, J., Lewis, K.A., 2010. Greenhouse gas emissions and energy use in UK-grown shortday strawberry (*Fragaria xananassa* Duch) crops. J. Agric. Sci. 148, 667–681.
Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the Environmental Burdens and Resource Use in the Production of Agricultural and Horticultural Commodities (Defra Project Report IS0205). Cranfield University and Defra, Bedford, p. 97.
Williams, A., Audsley, E., Sandars, D., 2010. Environmental burdens of producing bread wheat, oilseed rape and potatoes in England and Wales using simulation and system modelling. Int. J. Life Cycle Assess. 15, 855–868.
Zafiriou, P., Mamolos, A.P., Menexes, G.C., Siomos, A.S., Tsatsarelis, C.A., Kalburtji, K.L., 2012. Analysis of energy flow and greenhouse gas emissions in organic, integrated and conventional cultivation of white asparagus by PCA and HCA: cases in Greece. J. Clean. Prod. 29–30, 20–27.
Zehetmeier, M., Baudracco, J., Hoffmann, H., Heißenhuber, A., 2012. Does increasing milk yield per cow reduce greenhouse gas emissions? A system approach. Animal 6, 154–166.