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Does organic farming reduce environmental impacts?

–A Meta-Analysis of European research

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Abstract

Organic farming practices have been promoted as, *inter alia*, reducing the environmental impacts of agriculture. This meta-analysis systematically analyses published studies that compare environmental impacts of organic and conventional farming in Europe. The results show that organic farming practices generally have positive impacts on the environment per unit of area, but not necessarily per product

unit. Organic farms tend to have higher soil organic matter content and lower nutrient losses (nitrogen leaching, nitrous oxide emissions and ammonia emissions) per unit of field area. However, ammonia emissions, nitrogen leaching and nitrous oxide emissions per product unit were higher from organic systems. Organic systems had lower energy requirements, but higher land use, eutrophication potential and acidification potential per product unit. The variation within the results across different studies was wide due to differences in the systems compared and research methods used. The only impacts that were found to differ significantly between the systems were soil organic matter content, nitrogen leaching, nitrous oxide emissions per unit of field area, energy use and land use. Most of the studies that compared biodiversity in organic and conventional farming demonstrated lower environmental impacts from organic farming. The key challenges in conventional farming are to improve soil quality (by versatile crop rotations and additions of organic material), recycle nutrients and enhance and protect biodiversity. In organic farming, the main challenges are to improve the nutrient management and increase yields. In order to reduce the environmental impacts of farming in Europe, research efforts and policies should be targeted to developing farming systems that produce high yields with low negative environmental impacts drawing on techniques from both organic and conventional systems.

Keywords: biodiversity, greenhouse gas emissions, nitrogen leaching, soil organic matter, life cycle assessment

1 Introduction

Organic farming is often perceived to have generally beneficial impacts on the environment compared to conventional farming (Aldanondo-Ochoa and Almansa-Sáez, 2009; Gracia and de Magistris, 2008). Organic farming is regulated internationally by *Codex Alimentarius* Guidelines (established by The United Nations' Food and Agricultural Organisation (FAO) and the World Health Organisation) and by the International Federation of Organic Agriculture Movements' (IFOAM) Basic Standards. The latter are based on four principles (IFOAM, 2008): i) health: organic agriculture is intended to produce high quality food without using mineral fertilisers, synthetic pesticides, animal drugs and food additives that may have adverse health effects, ii) ecology: organic agriculture should fit the cycles and balances in nature without exploiting it by using local resources, recycling, reuse and efficient management of materials and energy, iii) fairness: organic agriculture should provide good quality of life, contribute to food sovereignty, reduce poverty, enhance animal well-being and take future generations into account, iv) care: precaution and responsibility have to be applied before adopting new technologies for organic farming and significant risks should be prevented by rejecting unpredictable technologies, such as genetic engineering.

Codex Alimentarius Guidelines and IFOAM Basic Standards provide a minimum baseline for national and regional standards worldwide. National standards take the local conditions into account and tend to be stricter than the IFOAM Basic Standards. In the European Union (EU), organic farming is regulated according to the European

Council Regulation No 834/2007 (EC, 2007), which sets the basis for national standards in the EU. All organic producers are inspected by organic inspection bodies, which may be private or managed by government. In many countries private certification bodies have their own stricter standards than national standards require. In this paper, organic farming was regarded as farming that is certified according to national standards. Conventional farming in this paper, describes non-organic farming systems that use pesticides and synthetic fertilisers and other farming practices that are regionally typical.

A range of different approaches has been used in order to compare environmental impacts of organic and conventional farming systems. Some studies have focused only on a particular aspect, for example biodiversity (Bengtsson et al., 2005; Feber et al., 2007; Fuller et al., 2005; Hole et al., 2005; Rundlof et al., 2008), land use (Badgley et al., 2007), soil properties (Maeder et al., 2002; Stockdale et al., 2002) or nutrient emissions (Syväsalo et al., 2006; Trydeman Knudsen et al., 2006). Some review studies have assessed the overall contribution of organic farming by combining the research from various impact categories (Gomiero et al., 2008; Hansen et al., 2001; Pimentel et al., 2005). Life Cycle Assessment (LCA) studies have used a product approach to assess the environmental impacts of a product from input production up to the farm gate (Cederberg and Mattsson, 2000; Thomassen et al., 2008). Mondelaers *et al.* (2009) used a meta-analysis to compare the environmental impacts of organic and conventional farming including studies from around the world, examining land use efficiency, organic matter content in the soil, nitrate and

phosphate leaching to the water system, greenhouse gas (GHG) emissions and biodiversity.

The aim of this current study was to systematically review and analyse the studies comparing environmental impacts of organic and conventional farming in Europe. A meta-analysis was used to evaluate the results of peer-reviewed studies comparing the nutrient losses, biodiversity impacts, greenhouse gas (GHG) emissions, eutrophication potential, acidification potential, energy use and land use in organic and conventional farming systems in Europe. Aside from the different geographical focus, this study extends the work of Mondelaers *et al.* (2009) in covering a larger literature and in extending the coverage to include ammonia emissions, phosphorus emissions, eutrophication and acidification potential and energy use. In addition to comparing the environmental impacts of the systems, this paper also analyses the reasons for the differences between the systems and the reasons for the variation of the results across different studies.

2 Methods

2.1 Literature search

A systematic literature search was performed to find studies comparing environmental impacts of organic and conventional farming in Europe. The ISI Web of Knowledge (www.isiwebofknowledge.com) database was used. The search was performed on 26th September 2009 with no restriction on publication year. The following search term combinations were used: (organic AND conventional AND farming) OR (organic

AND conventional AND agriculture). The preliminary search was refined to the subject areas “agriculture”, “plant sciences”, “environmental sciences & ecology” and “biodiversity & conservation”. The document type was defined as “article” and language as “English”. The search resulted in a list of 644 references. First the potential papers were selected based on the title and abstract. This resulted in a list of 275 papers. Finally the full papers were inspected and the papers included in this study were selected based on the following criteria: i) the study was related to European farming systems, ii) the study compared organic and conventional farming and provided quantitative results on at least one of the following aspects: soil organic carbon, land use, energy use, GHG emissions, eutrophication potential, acidification potential, nitrogen leaching, phosphorus losses, ammonia emissions or biodiversity, and iii) the paper was published in a scientific peer-reviewed journal. All types of studies (i.e. original field investigations, modelling studies and life cycle assessment studies) were included in the study. This filtering resulted in 71 papers that were used in the meta-analysis and 38 papers that provided data for the biodiversity review.

2.2 Selection of indicators and data extraction

A range of indicators was selected in order to include all important environmental impact categories and also to compare different allocation methods and research approaches. The indicators were grouped to Life Cycle Assessment (LCA) and non-LCA indicators. LCA indicators were those where all impacts occurring during the production chain from input production up to the farm gate were taken into account (e.g. Thomassen *et al.*, 2008), whereas a non-LCA indicator takes into account only the impacts occurring directly from the farming processes. LCA indicators generally

aim at describing the magnitude of the final impact that may be caused by many pollutants, whereas non-LCA impacts in this study are only emissions of particular pollutants. LCA studies generally present the results by allocating the impacts per unit of product and per unit of field area, whereas non-LCA studies generally report results only per unit of field area.

From the 71 studies providing data for the meta-analysis, 170 cases were extracted, since each study generally provided results from multiple farming systems (e.g. arable farming and horticultural farming or, in a LCA study, different products). These cases provided 257 quantitative measures of the environmental impacts of organic and conventional farming. The studies included are presented in the Supplementary material (Table S1). Ten indicators were used to compare the environmental performance of the two systems (Table 1). Impacts were either reported per unit of field area, or per unit of product. In order to compare the impacts between the systems and explain the reasons for the differences, both quantitative and qualitative data were extracted from the studies.

In addition to the results, more data about each case were extracted, including detailed farming practices (e.g. fertilisation, organic matter inputs, crop rotation, crops, livestock and yields), location of the farm, type of the farm (experimental or commercial), number of farms and sample size. The studies were also grouped as either experimental or modelling studies. A study was regarded as a modelling study if the results were based even partly on secondary data or modelling instead of on direct experimental data. LCA studies were automatically considered as modelling

studies as some secondary data for the processes before the farm stage are always used.

Table 1. *Indicators and allocation units used in the meta-analysis. Impacts were allocated per unit of 'field area' or per unit of 'product' (e.g. per tonne of wheat or pork).*

	Allocation of impacts per unit of:	
	Field area	Product
1) soil organic matter	√	
2) nitrogen leaching	√	√
3) nitrous oxide emissions	√	√
4) ammonia emissions	√	√
5) phosphorus losses	√	
6) land use		√
7) energy use		√
8) greenhouse gas emissions		√
9) eutrophication potential		√
10) acidification potential		√
11) biodiversity	√	

Non-LCA indicators:

Soil organic matter (SOM). The impact category SOM was chosen in order to describe the level of soil quality. SOM was chosen because it has a positive impact on many other soil quality aspects, such as structure, erosion control, water retention and long term productivity and, therefore, it is a sufficient indicator for describing the overall level of soil quality (Shepherd et al., 2002). High soil carbon content is also beneficial in terms of sequestration of carbon (King et al., 2004). The correlation between relative organic matter inputs and the proportion of leys in the rotation to the SOM response ratios was also assessed. All studies included in the meta-analysis were field experiments.

Nitrogen leaching. Nitrogen leaching causes contamination of ground water, eutrophication of waterways and also, indirectly, nitrous oxide emissions. Nitrogen leaching occurs when more nitrate is available in the soil than plants can use at times of the year when water from rain, irrigation or snowmelt moves through the soil into the groundwater. The level of nitrogen leaching is influenced by soil and weather conditions and management decisions such as choice of crop rotation; type, timing and amount of nitrogen fertiliser application and type and timing of cultivation (Shepherd et al., 2003). Both modelling and field experiment studies were included in this study and the differences between these two groups were analysed.

Nitrous oxide emissions. Nitrous oxide is a greenhouse gas and therefore contributes to climate change. Nitrous oxide from agriculture originates mainly from application of nitrogen fertilisers, manure and nitrogen fixing crops. Nitrous oxide is produced in soils aerobically during nitrification and anaerobically during denitrification. Direct emissions arise from nitrogen applications (e.g. mineral fertilisers and manure) into the soil. Nitrous oxide is emitted indirectly through volatilisation and atmospheric deposition of ammonia and oxides of nitrogen that originate from the nitrogen applications into the soil and surface runoff and leaching of nitrogen from these sources (US-EPA, 2006). Most of the studies were field experiments and only few modelling studies were included. Therefore, the results of the experimental and modelling studies were not analysed separately.

Ammonia emissions. Ammonia emissions from agriculture arise mainly from manure. Ammonia is the main acidifying pollutant from agriculture. Ammonia is produced when urea in urine and manure comes into contact with the enzyme urease, which is found commonly in bacteria and fungi inhabiting manure and soil. Thus, animal housing, manure stores and the spreading of manure to land are the major sources of ammonia. Between 60-80% of the nitrogen fed to cattle is excreted, mainly as urine, and much of it is rapidly converted to ammonia (Webb et al., 2006). Most of the studies were field experiments and only few modelling studies were included. Therefore, the results of the experimental and modelling studies were not separately analysed.

Phosphorus losses. Phosphorus losses contribute to eutrophication of waterways. Many soils have large reserves of phosphorus, but often only one percent is available to crops (Shenoy and Kalagudi, 2005). Phosphorus is added to soils as phosphate fertilisers, and recycled back into the soil from plant residues, agricultural wastes and sewage sludge. Phosphate reserves are a non-renewable resource and the accelerating use of this raw material will eventually lead to depletion. The existing stock is estimated to sustain production for approximately 60 more years (Franz, 2008). Therefore, enhanced recycling of phosphorus, minimising phosphorus losses and better utilisation of soil phosphorus reserves are essential.

Biodiversity. The main reason for the loss of rural biodiversity during the last decades has been the change in agricultural landscapes (Gabriel et al., 2005; Luoto et al., 2003; Macdonald et al., 2000; Macdonald and Smith, 1991). In Europe, formerly

heterogeneous landscapes with a mix of small arable fields, semi-natural grasslands, wetlands and hedgerows have been replaced in many places with largely homogenous areas of intensively cultivated fields (Benton et al., 2003). This has resulted in declines in many animal and plant populations and caused a severe loss of species (Krebs et al., 1999). As the impacts of organic farming on biodiversity have been reviewed earlier by Hole et al. (2005) and Bengtsson *et al.* (2005) only the new literature (published after 2003) concerning biodiversity was reviewed in this study, and the results were combined and compared with the results of Hole et al. (2005). The included studies are listed in the Supplementary material (Table S2).

LCA indicators:

Land use. Land is a limited resource. The growing world population increases the pressure for land conversion resulting in the decrease of many natural habitats. It is estimated that a further 10.0 million km^2 of natural ecosystems will be converted to agriculture by 2050 if the current consumption trends continue (Tilman et al., 2001). The impact category 'land use' in this study relates to the area of land required for producing a unit of product output. Especially in organic systems, crop yield levels alone do not provide sufficient information about the total land requirements, since additional land is required for fertility building crops. Therefore, only studies using an LCA method were accepted for the land use category. Yield data were also collected, but they are presented separately from the land use assessment.

Energy use. Energy is used on farms directly in electricity and fuel oils, and indirectly in the manufacture and transport of fertilisers, pesticides, animal feeds and in the manufacturing and maintenance of machinery. The production and distribution of mineral fertilisers account for 37% of the total energy input of an agricultural product and the production of pesticides accounts for approximately 5% (Deike et al., 2008). For energy use only studies comparing the total direct and indirect energy requirement per product unit during the production chain from input production up to the farm gate were accepted.

Greenhouse gas (GHG) emissions. GHG emissions were measured as carbon dioxide equivalents (CO_2-eq). The major GHG emissions from agriculture are carbon dioxide, methane and nitrous oxide. GHG emissions factors over a 100-year time frame were used, as all of the studies reviewed reported the results by using this time frame. There is a significant relationship between total nitrogen input and nitrous oxide emissions (Jones et al., 2007; Petersen et al., 2006b). The main agricultural methane sources globally are enteric fermentation of ruminant livestock (55%), stored manures (13%), rice grown under flooded conditions (19%) and land use change (including burning of biomass) (13%), (US-EPA, 2006). Agriculture is both a source and a sink for carbon dioxide (CO_2) (Smith et al., 2008). Sources include the use of fossil fuels in machinery and production of farming inputs, such as fertilisers and pesticides. Organic matter, which can act as a temporary store for atmospheric carbon, acts as a CO_2 sink in agriculture.

Eutrophication potential. Eutrophication means enrichment of terrestrial and aquatic habitats with plant nutrients, which results in increased growth of plants and algae. The main agricultural sources are nitrate, phosphate and ammonia. Eutrophication potential is quantified in terms of phosphate equivalents (Huijbregts and Seppälä, 2001). Aquatic eutrophication occurs due to the enrichment of waterways with plant nutrients, mainly phosphorus and nitrogen, which results in the increased production of aquatic plants and algae. This can cause fish kills, harm wildlife, and impair the water use for recreation, industry and drinking. Agriculture is the main contributor to eutrophication accounting 50-80% of the total aquatic nitrogen load and about 50% of the phosphorus load in Europe (EEA, 2005). Nitrogen is more commonly the key limiting nutrient of marine waters, while phosphorus is the limiting factor in freshwaters (Smith et al., 1999).

Acidification potential. The major acidifying pollutants from agriculture are ammonia (NH_3) and sulphur dioxide (SO_2). Acidification potential is quantified in terms of SO_2 equivalents (SO_2-eq) (Seppälä et al., 2006). Acidifying pollutants impact on soil, ground- and surface waters, biological organisms and other materials, for example, causing fish mortality, forest decline and the erosion of buildings. Agriculture, particularly livestock production, accounts for about 80% of ammonia emissions in Europe (EMEP, 2008). Ammonia was predicted to be the largest source of acidifying gases in Europe by 2010 (Webb et al., 2006).

2.3 Data analysis

The response ratios for each indicator were calculated using the following formula:

Response ratio = [(impact of organic farming / impact of conventional farming) – 1]. Thus, negative values indicate lower impacts from organic farming and positive values indicate higher impacts from organic farming relative to conventional farming. The median values of the response ratios for each impact category were calculated. Results were not weighted according to the sample size, therefore all cases had equal impact on the results and the N values represent the number of cases.

The analytical approach was adapted from Benayas et al. (2009). The normality of the data was tested by using the Kolmogorov-Smirnov test. Not all impact ratios were normally distributed, therefore a Wilcoxon Signed Rank test was used to determine whether the median impact ratios were significantly different from zero. The correlations between some farming practices and environmental impacts were examined using the Spearman Rank test. SPSS 14.0 software was used for the statistical analyses.

3 Results and discussion

3.1 Soil Organic Matter (SOM)

The median SOM content across all the cases was 7% higher in organic farms compared to conventional farms. Regardless of a wide variation, the median difference between the systems was statistically significant (Figure 1A).

The main explanation for higher organic matter contents in organic systems was that organic systems had higher organic matter inputs. In the systems included in this

meta-analysis, the organic matter inputs in the form of manure or compost were on average 65% higher on organic farms compared to conventional farms, when calculated as an average of the relative inputs by weight (organic/conventional) in each case. There was no correlation between organic matter inputs and SOM (Spearman's Rho = 0.24, N = 46; P = 0.11). Therefore the organic matter inputs alone cannot explain the differences in SOM content levels between the systems. Organic farms in the data had on average 14% more leys in the rotation although there was no correlation between ley area and SOM content (Spearman's Rho = 0.08, N = 47, P = 0.58).

Some cases reported a higher SOM content in organic systems even though the organic matter inputs were similar in the systems compared. It was suggested that mineral nitrogen additions satisfy the nitrogen requirements of micro-organisms, and therefore, increase the decomposition rate of organic residues (Birkhofer et al., 2008). Other explanations for higher SOM levels in organic systems were less intensive tillage and inclusion of leys in the rotation (Canali et al., 2009; Cardelli et al., 2004; Quintern et al., 2006).

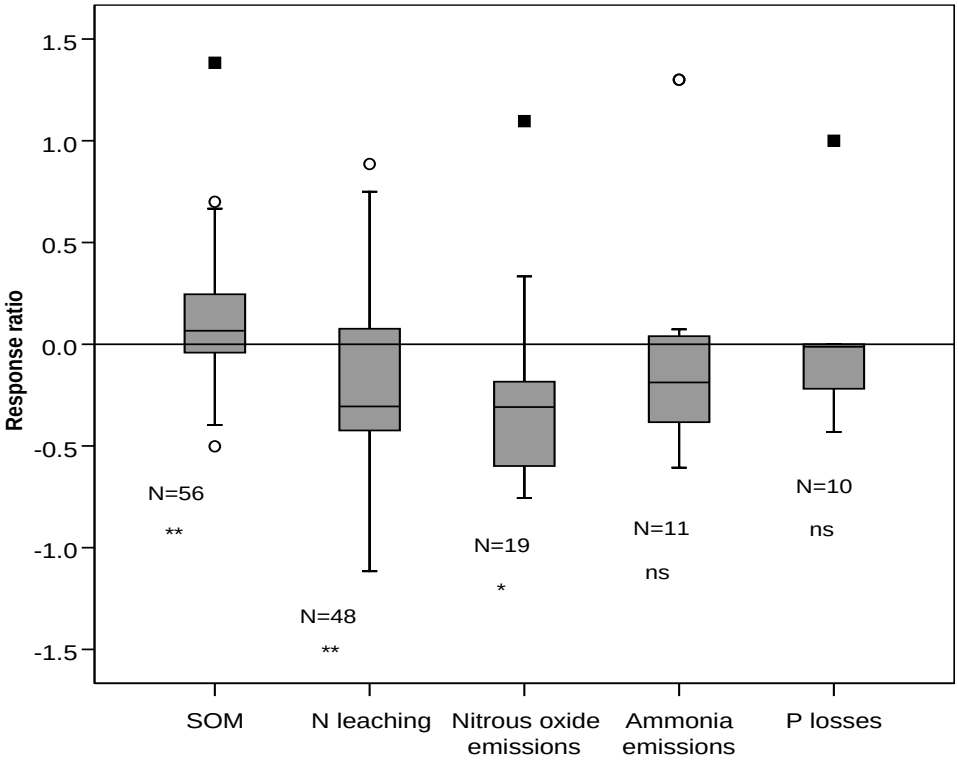
Gosling and Shepherd (2005) explained observed lower SOM contents in organic systems by higher yields, and therefore, higher crop residue leftovers in conventional systems, which can compensate the lower external organic matter inputs. They also argued that leys do not necessarily contribute to the increase of SOM, because they have a low carbon-nitrogen ratio and therefore organic matter decays rapidly. Furthermore, they pointed out that intensive tillage needed for incorporation of leys is

likely to offset the benefits gained while the ley is in place. In addition, some studies suggested that the number of years the field has been under organic management (Marinari et al., 2006) and the time of soil sampling (Monokrousos et al., 2008) have an impact on the results. However, the meta-analysis data did not show a correlation between duration of organic farming and the relative SOM content (Spearman's Rho = -0.09, N = 42, P = 0.56).

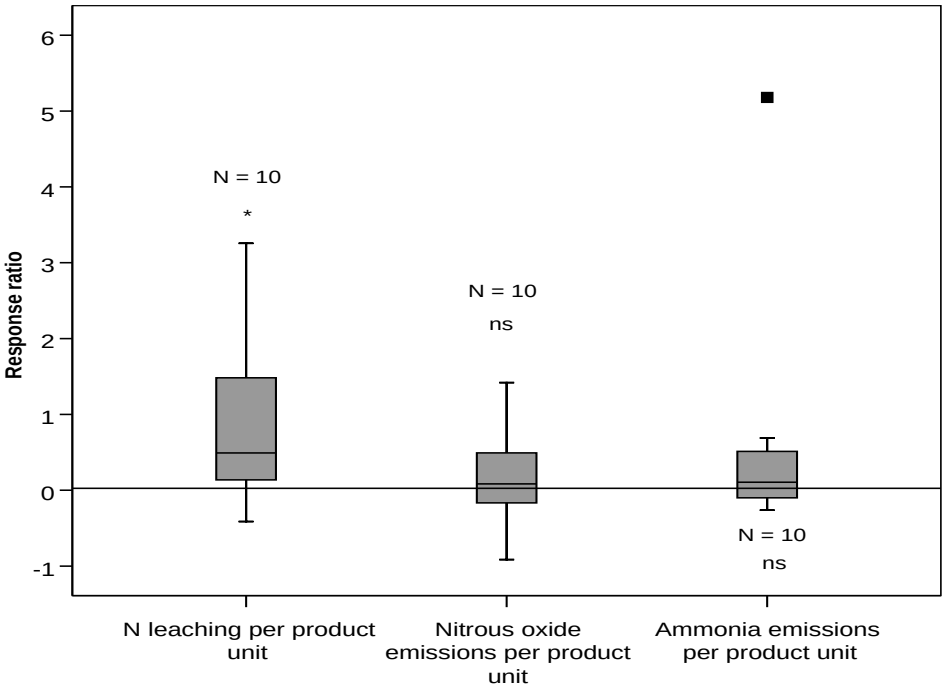
In order to analyse the data further, the cases were grouped in different categories based on the manure input: i) organic less manure than conventional, ii) organic more manure, and iii) both systems with the same amount of manure input. When the organic system had more or the same amount of manure input than conventional system, the SOM content in organic systems was significantly higher (Figure 2). When organic systems had lower manure input than conventional systems, the difference in SOM content between the systems was not significant. However, as SOM content is impacted by the soil management history, some comparative studies may give misleading results unless soils with same management history are compared.

The results indicate that organic farming generally leads to higher SOM content, but some conventional farming systems do have the potential to achieve similar or even higher SOM levels when they include the application of manures.

A Non-LCA impacts per unit of land



B Non-LCA impacts per unit of product



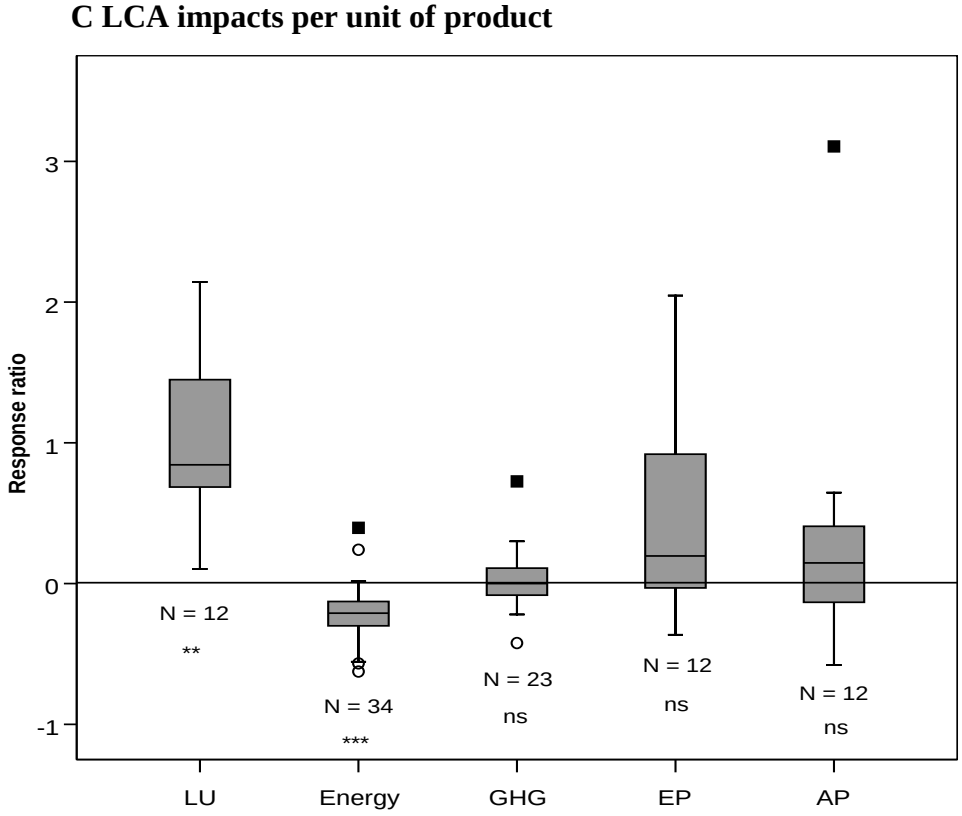


Figure 1. *Response ratios for **A)** non-LCA (Life Cycle Assessment) impacts allocated per unit of land: soil organic matter (SOM), phosphorus (P) losses, nitrogen (N) leaching, nitrous oxide emissions and ammonia emissions, and **B)** non-LCA impacts allocated per unit of product: nitrogen (N) leaching, nitrous oxide emissions and ammonia emissions, and **C)** LCA impacts allocated per unit of product: energy use, greenhouse gas emissions (GHG), acidification potential (AP), eutrophication potential (EP) and land use (LU). (Line through the box: median; upper and lower sides of the boxes: upper and lower quartiles; tiles: extreme values; O, ■ : outliers 1.5-3 and over 3 box lengths from the upper or lower edge of the box, respectively; positive values: impacts from organic farming are higher, negative values: impacts from organic farming are lower, N=number of cases, ns=not significantly different from zero (Wilcoxon Signed Rank test P>0.05); , ***P<0.001; **P<0.01; *P<0.05; Z-values: SOM = -3.078, N leaching = -2.845, Nitrous Oxide emissions = -2.575, Ammonia emissions = -0.801, P losses = -0.943, N leaching per product unit = -2.395, Nitrous oxide emissions per product unit = -0.255, Ammonia emission per product unit = -1.362, LU = -3.059, Energy = -4.214, GHG = -0.373, EP = -1.726 and AP = -1.177)*

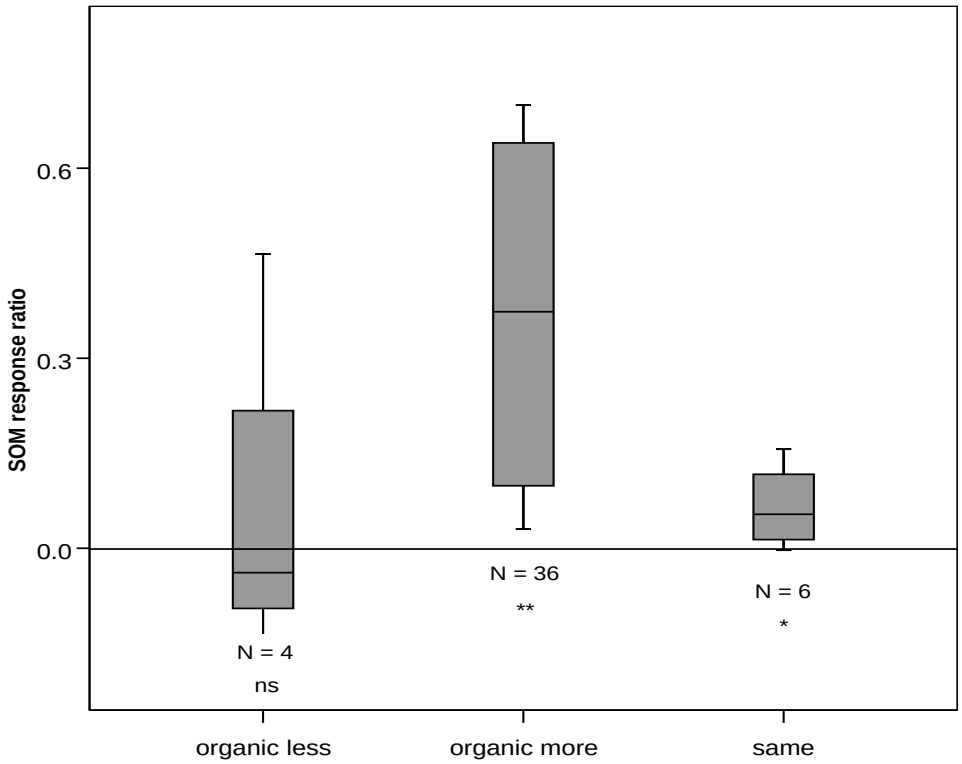


Figure 2. *Response ratios for soil organic matter (SOM) when cases are grouped based on the relative manure inputs (manure input by weight in an organic system divided by the input in a conventional system) between the systems. (Line through the box: median; upper and lower sides of the boxes: upper and lower quartiles; tiles: extreme values; positive values: impacts from organic farming are higher, negative values: impacts from organic farming are lower; N=number of cases in the sample; ns=not significantly different from zero (Wilcoxon Signed Rank test P>0.05*); ***P<0.01; *P<0.05; Z-values Z-values (respectively from left to right on the graph): -0.365, -3.111 and -1.992)*

3.2 Nitrogen and phosphorus losses

Median response ratios showed that nitrogen leaching per unit of area was 31% lower from organic farming compared to conventional farming and 49% higher per unit of product (Figures 1A and 1B). The median response ratio of N leaching per unit of area based on original field investigations alone (-0.1053) was not significantly different from zero, whereas the median response ratio of model based studies (-0.4032) was significantly different from zero (Figure 3). There was a large but insignificant correlation between nitrogen input ratio and nitrogen leaching in the

'field experiments' group (Spearman's Rho = -0.80, N = 5, P = 0.10), whereas there was no correlation in the 'models' group (Spearman's Rho = -0.07, N = 31, P = 0.71). There was no correlation between the proportion of grass in the rotation and nitrogen leaching response ratio (Spearman's Rho = -0.13, N = 18, P = 0.60). These results suggest that modelling studies may overestimate the benefits of organic farming in the reduction of nitrogen leaching.

The main explanation for lower nitrogen leaching levels from organic farming per unit of area was the lower levels of nitrogen inputs applied (Hansen et al., 2000; Korsaeth, 2008; Torstensson et al., 2006; Trydeman Knudsen et al., 2006). Higher nitrogen leaching levels were explained by poor synchrony between the nutrient availability and crops' nutrient intake (Aronsson et al., 2007). Especially, after incorporation of leys, the nitrogen losses tend to be high (Syväsalo et al., 2006). The use of cover crops in conventional systems was found to reduce nitrogen leaching resulting in a lower leaching level than organic farming (Torstensson et al., 2006).

Median nitrous oxide emissions were 31% lower from organic systems when the impact was allocated per unit of field area, but 8% higher when the impact was allocated per unit of product (Figures 1A and 1B). Median ammonia emissions followed a similar trend with organic systems having 18% lower emissions per unit of area and 11% higher per unit of product (Figures 1A and B). The lower nitrous oxide and ammonia emissions from organic farming per unit of area were mainly due to lower overall nitrogen inputs in organic than in conventional systems.

The median response ratio for phosphorus losses showed 1% lower emissions from organic systems (Figure 1A). Only one study found lower phosphorus losses from a conventional system (Aronsson et al., 2007). That was due to incorporation of green manure resulting in increased mineralisation of crop residues in the organic system. The organic systems included in the study had 55% lower total phosphorus inputs compared to conventional systems.

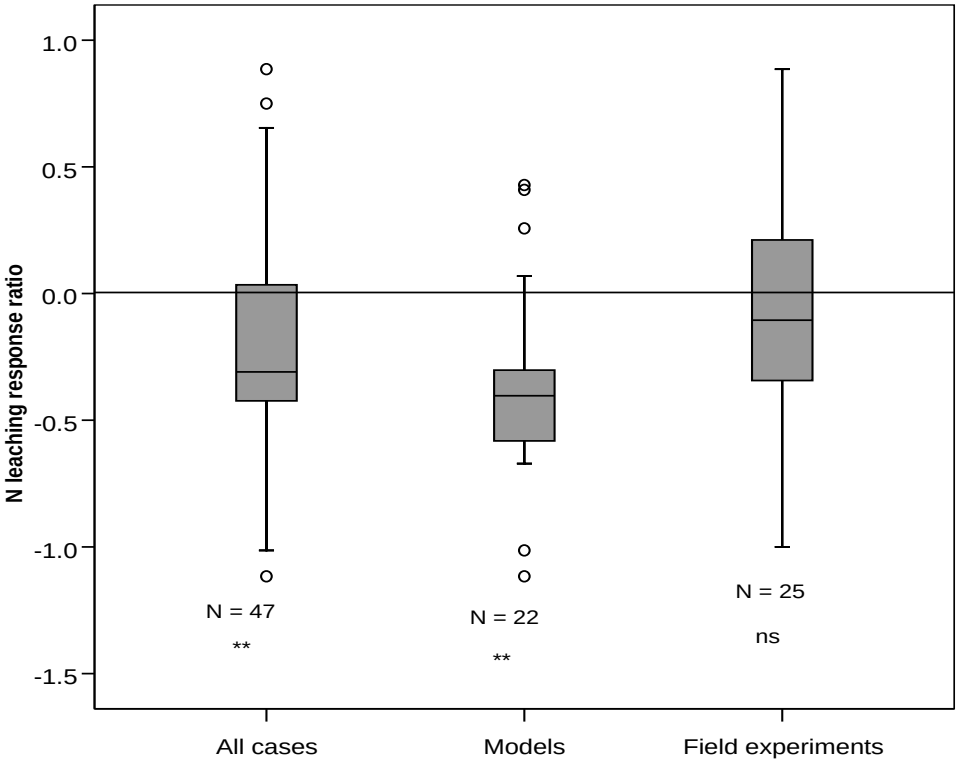


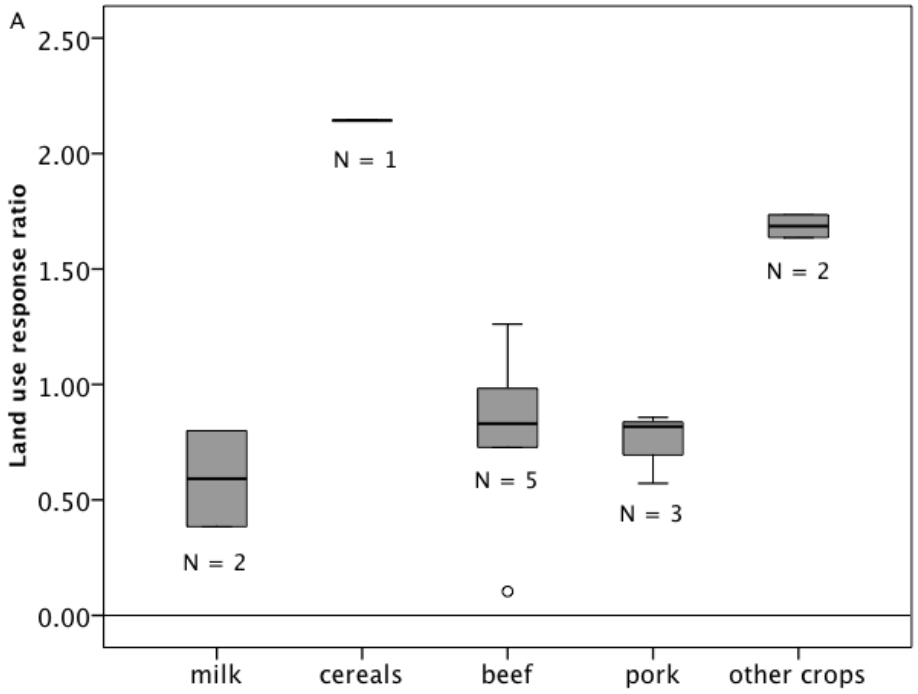
Figure 3. *Response ratios for nitrogen leaching per unit of land area grouped by the type of study (field experiment or model). (Line through the box: median; upper and lower sides of the boxes: upper and lower quartiles; tiles: extreme values; O : outliers 1.5-3 box lengths from the upper or lower edge of the box, respectively; positive values: impacts from organic farming are higher, negative values: impacts from organic farming are lower; N=number of cases in the sample; ns=not significantly different from zero (Wilcoxon Signed Rank test P>0.05*), ***P<0.01; *P<0.05*; *Z-values: All cases = -2.912, Field experiments = -1.019, Models = -3.007)*

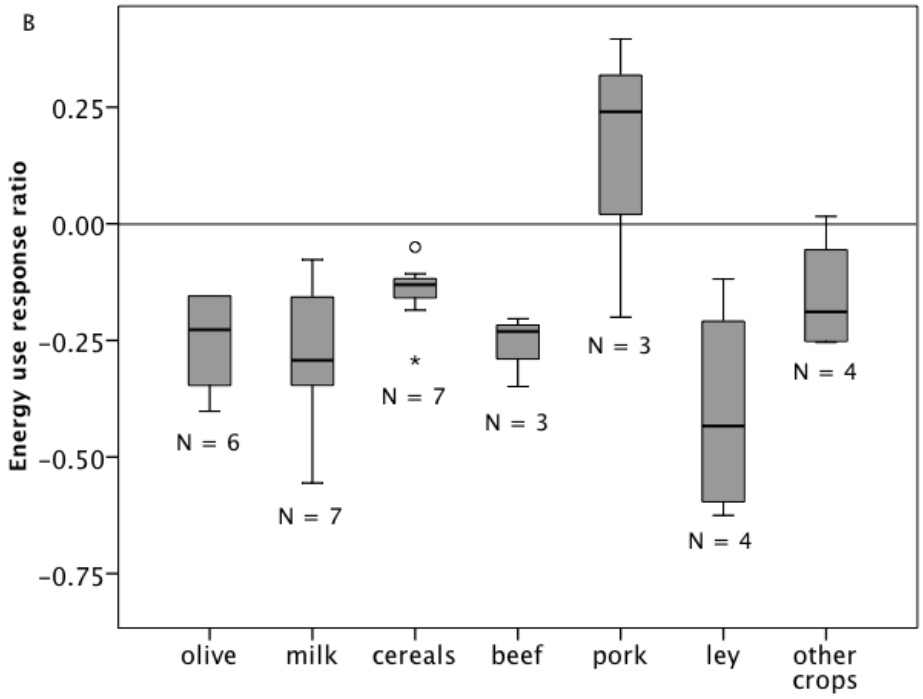
3.3 Land use

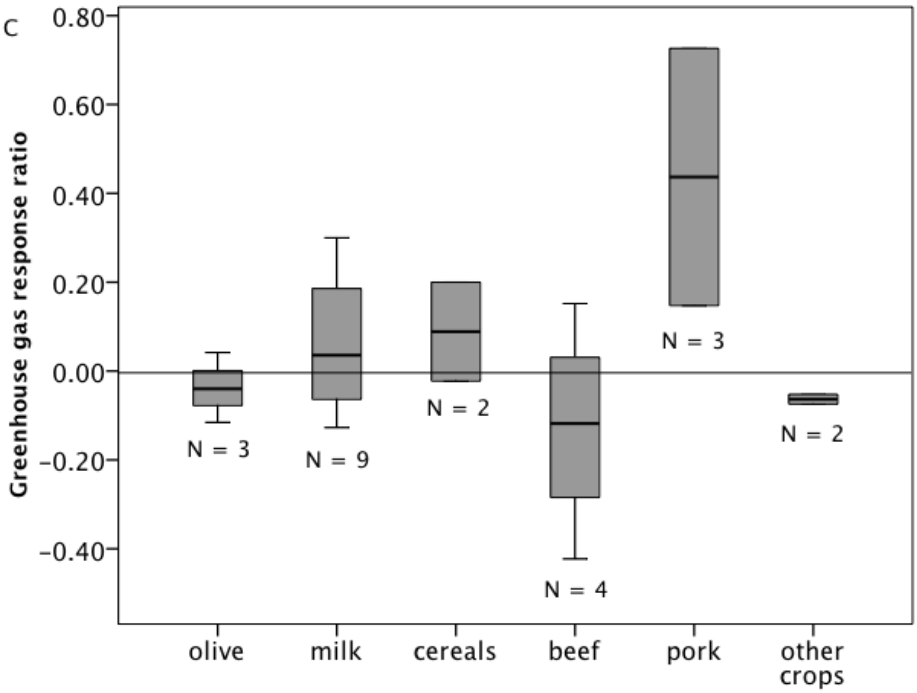
Median response ratio showed that organic farming requires 84% more land compared to conventional farming in Europe (Figures 1C and 4A). This is mainly due to lower crop yields, lower yielding animals and land area requirement for fertility building crops. The average organic yields over all crops in the data were 75% (SD ±17%) of conventional yields (Table 2). In only in two cases were the organic yields higher, both of which related to melon production (Melero et al., 2006). The main reason for lower organic yields identified in the studies was insufficient availability of nutrients (especially nitrogen) although some studies mentioned problems with weeds, diseases or pests (Korsaeth, 2008). The higher organic melon yields were explained by higher SOM content in the organic fields compared to the conventional fields (Melero et al., 2006). The studies that found similar yield levels from both systems compared yields in experimental farms that had high soil quality (Herencia et al., 2008; Melero et al., 2006).

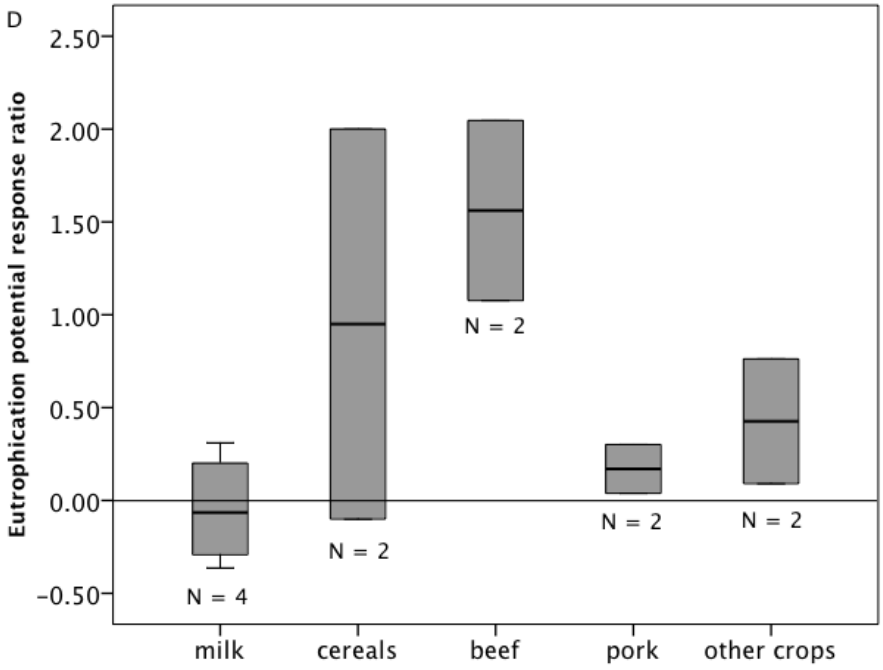
Table 2. *Relative minimum, maximum and mean yields (organic/conventional), standard deviation of the means (SD) and number of cases (N).*

	N	Min	Max	Mean	SD
Winter wheat	16	0.41	0.86	0.62	0.12
Spring wheat	5	0.70	0.87	0.78	0.06
Barley	14	0.25	0.85	0.65	0.18
Oat	5	0.40	0.80	0.61	0.17
Other cereals	4	0.48	0.83	0.67	0.15
Potato	11	0.17	1.32	0.68	0.37
Vegetables	13	0.60	1.00	0.79	0.16
Sugar beet	2	0.76	1.11	0.94	0.25
Leys	20	0.65	1.10	0.85	0.11
Olive	1	0.68	0.68	0.68	0.00
Citrus	1	0.68	0.68	0.68	0.00
Melons	2	1.64	1.81	1.73	0.11
Oilseed rape	2	0.53	1.11	0.82	0.40


A
Land use response ratio
2.50
2.00
1.50
1.00
0.50
0.00
N = 1
N = 2
N = 5
N = 3
N = 2
milk
cereals
beef
pork
other crops


B
Energy use response ratio
0.25
0.00
-0.25
-0.50
-0.75
N = 6
N = 7
N = 7
N = 3
N = 3
N = 4
N = 4
olive
milk
cereals
beef
pork
ley
other crops


C
Greenhouse gas response ratio
0.80
0.60
0.40
0.20
0.00
-0.20
-0.40
N = 3
N = 9
N = 2
N = 4
N = 3
N = 2
olive
milk
cereals
beef
pork
other crops


D
Eutrophication potential response ratio
2.50
2.00
1.50
1.00
0.50
0.00
-0.50
N = 4
N = 2
N = 2
N = 2
N = 2
milk
cereals
beef
pork
other crops

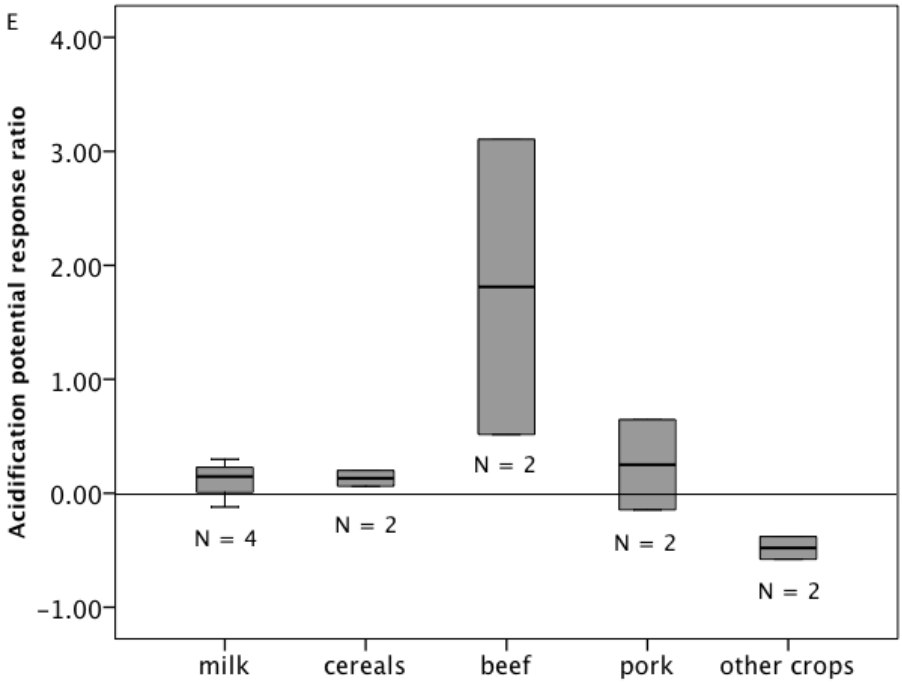


Figure 4. *Response ratios for energy use (A), greenhouse gas emissions (B), acidification potential (C), eutrophication potential (D) and land use (E) grouped by product. (Line through the box: median; upper and lower sides of the boxes: upper and lower quartiles; tiles: extreme values; O, * : outliers 1.5-3 and over 3 box lengths from the upper or lower edge of the box, respectively; positive values: impacts from organic farming are higher, negative values: impacts from organic farming are lower; N=number of cases in the sample)*

3.4 Energy use

Median energy use showed 21% lower energy consumption in organic farming systems per product unit (Figure 1C) although the variation was wide; from 63% lower energy use up to 40% higher energy use in organic systems. Only three cases out of 34 found higher energy use from organic systems of which two cases were pork production (Basset-Mens and van der Werf, 2005) and one potato production (Glendining et al., 2009) (Figure 4B). Higher energy inputs in conventional farming

were mainly due to the high energy needed for production and transport of non-organic fertilisers, especially synthetic nitrogen fertilisers.

3.5 Greenhouse gas emissions

The median response ratio for GHG emissions was zero (Figure 1C). There were clear differences in the median response ratios between different product groups (Figure 4C). Organic olive, beef and some crops had lower GHG emissions whereas organic milk, cereals and pork had higher GHG emissions compared to conventional products. The emissions from olive production were closely related to the amount of fossil fuels used and that was found to be independent of the farming system (Kaltsas et al., 2007). In most of cases organic milk production had higher GHG emissions compared with conventional systems. Higher GHG emissions in organic systems were due to higher methane and nitrous oxide emissions and lower milk production per animal (Thomassen et al., 2008). Only Cederberg & Mattson (Cederberg and Mattsson, 2000) and one of the cases in Olesen *et al.* (2006) found lower GHG emissions from organic milk production. Organic beef production was found to have lower GHG emissions compared to conventional due to lower emissions from industrial inputs (Casey and Holden, 2006). GHG emissions from organic pork production were higher, because of high nitrous oxide emissions from straw litter.

3.6 Eutrophication and Acidification Potential

The median response ratio for eutrophication potential was 0.196 (Figure 1C). These results were skewed by Thomassen *et al.*'s (2008) results that showed 36% lower

eutrophication potential in organic milk production, but this may have been caused by different soil types on organic and conventional farms. Higher eutrophication potential for organic products was mainly due to lower animal and crop yields. Eutrophication potential per unit area was generally lower in organic systems due to their lower nutrient inputs, but higher per product unit due to lower animal and crop yields as compared to conventional systems. The comparison between the products showed that organic milk tended to have lower eutrophication potential than conventional milk, whereas in the other product categories organic products had higher impact than conventional (Figure 4D).

The median response ratio for acidification potential was 0.147, whereas for ammonia emissions per unit of area it was -0.188. Lower ammonia emissions per unit of area were due to lower nitrogen inputs in organic systems, whereas the acidification potential per product unit was higher due to lower crop and animal yields. Cederberg & Mattson (2000) explained lower acidification potential from organic milk production to be due to the higher protein content in conventional feed and therefore higher ammonia emissions. Basset-Mens & van der Werf (2005) found lower acidification potential from organic pork production due to lower animal density in buildings, use of catch crops in winter and use of solid manure. However, they found that a conventional system using targeted practices for reducing ammonia emissions, such as better control of indoor climate, optimised design of buildings and covering slurry stores, had lower acidification potential than the organic system. When different products were compared, it was found that organic livestock products and

cereals had higher acidification potential, whereas some organic crop products had lower acidification potential than conventional products (Figure 4E).

3.7 Biodiversity

A meta-analysis that compared biodiversity in organic and conventional farms (Bengtsson et al., 2005), found organic farms generally to have 30% higher species richness and 50% higher abundance of organisms than conventional farms. However, it also showed a wide variation between different studies. Sixteen per cent of the studies found a negative effect of organic farming on species richness. The differences were more prominent at the plot scale than in matched landscapes. They also found that the effect of organic farming on species richness was larger in intensively managed landscapes than in diverse landscapes with many non-crop biotopes.

In a review of literature published between 1981 and 2003, Hole *et al*. (2005) compared biodiversity in organic and conventional farms. They found that organic farming generally had positive impacts on many species (Table 3). However, they concluded that it is still unclear whether conventional farming with specific practices for biodiversity conservation (i.e. agri-environmental schemes) can provide higher benefits than organic farming.

More studies published after 2003 supported the findings of Hole *et al*. (2005) and Bengtsson *et al*. (2005) (Table 3), but none found organic farming to have negative impacts on biodiversity. In particular, weed plant richness has been widely found to

be greater in organic farms compared with conventional farms (Albrecht, 2005; Gabriel et al., 2006; Petersen et al., 2006a; Romero et al., 2008; Roschewitz et al., 2005). Some studies showed that landscape had more significant impact on biodiversity than farming regime (Kragten and Snoo, 2007; Piha et al., 2007; Purtauf et al., 2005; Rundlöf et al., 2008; Rundlöf and Smith, 2006). It has also been found that organic farming alone without additional practices is not adequate for conserving some bird species (Kragten and Snoo, 2007; Piha et al., 2007) nor butterflies (Ekroos et al., 2008). The question as to whether conventional farming with specific targeted practices can result in higher biodiversity than organic farming is yet to be fully answered.

Table 3. Numbers *of studies showing positive, negative or mixed/no difference when species abundance and/or richness where compared in organic versus conventional farming. The data for the years 1981-2003 are from Hole et al. (2005).*

	Positive			Negative			Mixed/No difference		
Taxon	1981-2003	2004-2009	Total	1981-2003	2004-2009	Total	1981-2003	2004-2009	Total
Birds	7	3	10			0	2	2	4
Mammals	2	1	3			0			0
Butterflies	1	3	4			0	1	2	3
Spiders	7	1	8			0	3		3
Earthworms	7	1	8			0	4	2	6
Beetles	13	3	16	2		2	3	2	5
Other arthropods	7	3	10	5		5	2	2	4
Plants	13	10	23	1		1	2	1	3
Soil microbes	9	9	18		1	1	8	3	11
Total	66	34	100	8	1	9	25	14	39

4 Discussion

4.1 Comparison with a previous study

The results of this study are largely consistent with the findings of Mondelaers *et al*. (2009) (Table 4), although they also included non-European studies. The largest difference was in the results of nitrogen leaching per kg of product, where our study

showed 49% higher impacts from organic farming whereas Mondelaers et al. showed 5% lower impacts from organic system. This difference can be explained by the fact that our study included different studies from those included by Mondelaers et al., as non-European studies were excluded and more European studies were included in our study. We included 37 papers that they did not consider, partly reflecting the fact that they did not include some impact categories that were included in this study: ammonia emissions, phosphorus losses, acidification potential, eutrophication potential and energy use. It would be valuable in further work to explore whether there are systematic differences in the definitions and relationships considered between different regions of the world or between different categorisations of organic farming.

Table 4. Comparison of the results of this study with the results of Mondelaers et al. (2009)(both results presented as response ratios: (impact of organic farming / impact of conventional farming) – 1.

Impact category	**This study**	**Mondelaers et al.**
Yields	-0.22	-0.186
Soil organic matter	0.066	0.064
Nitrogen leaching (per ha)	-0.306	-0.297
Field experiment	-0.105	-0.260
Model	-0.403	-0.424
Nitrogen leaching (per kg)	0.491	-0.050
Greenhouse gas emissions (per ha)	not included	-0.392
Greenhouse gas emissions (per kg)	0.000	-0.101
Nitrous oxide emissions (per ha)	-0.309	-0.140
Nitrous oxide emissions (per kg)	0.085	not included
Ammonia emissions (per ha)	-0.188	not included
Ammonia emissions (per kg)	0.106	not included
Phosporus losses (per ha)	-0.013	not included
Eutrophication potential (per kg)	0.196	not included
Acidification potential (per kg)	0.147	not included
Energy use (per kg)	-0.211	not included

4.2 Opportunity costs of land use

As organic farming requires more land than conventional farming to produce a unit of product, large scale conversion to organic farming could provide environmental benefits only at the expense of reducing food production or extending production into other areas of land. Given that the demand of food is increasing globally, there will be pressures to increase rather than reduce production per unit area (The Royal Society, 2009). This emphasises the importance of analysing environmental impacts per unit of production rather than of area.

It is thus important to recognise the alternative land use options (Berlin and Uhlin, 2004; Tuomisto et al., 2012a). In order to assess the overall environmental performance of the different systems, the alternative uses of the extra land requirements should be taken into account. If less land was used for agriculture, more land could be used for other purposes such as wildlife conservation, biofuel production or forestry.

4.3 The challenges in organic farming systems

In our view, the main challenge for organic farming systems to improve overall sustainability is to increase yields without causing harm to the environment. The main reasons for low yields in organic farms are soil nutrient deficiencies and problems with pests, diseases and weeds (Köpke et al., 2008). Research experiments have shown that under carefully controlled management conditions organic farming has the potential to achieve yields comparable with those in conventional farming (Jonsson,

2004; Pimentel et al., 2005). However, these controlled conditions may be impossible to achieve on commercial farms. Further research is needed to improve control strategies for weeds, pests and diseases in organic systems, especially when reduced tillage is used. There is also a need for breeding both crops and animals that are suitable for organic farming, since many crops varieties and animal breeds currently used in organic farming have been developed for conventional farming systems (Wolfe et al., 2008).

Nutrient management in organic farming systems could be improved by incorporating anaerobic digestion technology (Stinner et al., 2008; Tuomisto and Helenius, 2008; Tuomisto et al., 2012a). Green-manure yield, cover crops and crop residues from organic crop farms, and manure from livestock farms, can be collected and treated in an anaerobic digester. The methane biogas produced can be used for heating, electricity production or engine fuel, and the digestate (the residue of the process) can be returned back to the fields as a fertiliser. Anaerobic digestion converts the organically bound nitrogen to ammonium form, and thus, more readily available for the crop (Lehtomäki, 2006). Biogas production enhances the energy balance of the farm and improves the nutrient management, because the nutrients can be applied into the soil when the crop's nutrient intake is highest. Nutrient cycling can be further enhanced by using raw materials from human communities, e.g. biowaste, within anaerobic digesters.

4.4 The challenges in conventional farming systems

The key challenges in conventional farming are to improve soil quality, reduce the use of pesticides and mineral fertilisers and enhance and protect biodiversity. We believe that an optimal outcome may be achieved by using an integrated farming approach, in which weed, pest and disease controls are based mainly on preventive practices, such as versatile crop rotation and use of clean seeds, and pesticides are used only if particularly advantageous. The management of soil quality is a high priority for maintaining the long term productivity. Therefore, practices that increase the soil organic matter content and the structure of soil should be used. These practices include crop rotations with crops that have large root mass, additions of organic material and reduced or no-tillage regimes. Further research is needed for investigating the possibilities of storing carbon in the soils and improving the soil properties by utilising biochar technology (Roberts et al., 2010).

As the production of mineral nitrogen fertilisers is highly energy demanding and phosphorus resources are diminishing, it may be argued that fertilisation should increasingly be based on biological nitrogen fixation, recycling of nutrients and efficient use of soil nutrient storage. Efficient recycling of nutrients also requires returning nutrients back to the fields from human communities as a form of biowaste or sewage sludge. Anaerobic digestion technology should be utilised in conventional farms in a similar way to that described earlier for organic farms (Section 4.3). Nutrient balances should be maintained by mixing livestock and crop farms at a regional spatial scale, so that manure is utilised efficiently without over fertilisation. Animal feeding should be based on forage, so that grasslands that are not suitable for

crop production are used for extensive animal grazing, whereas waste products from farming and food industries can be used for feeding pigs and poultry. Livestock breeding programmes should focus on reducing the nitrogen and methane emissions produced by livestock (Defra, 2008).

Plant breeding can offer many effective solutions towards more sustainable farming systems. In that respect, the use of GM technology may offer opportunities, because it has a potential to hasten the development of new beneficial varieties. GM crops can, for example, reduce the use of pesticides (Brookes, 2008), convert contaminated sites into safe agricultural land (Macek et al., 2008), or utilise soil nutrient sources more efficiently (Shenoy and Kalagudi, 2005). There are many concerns about the possible risks of GM technology to the environment and human health. However, it has not been shown that the process of genetic modification itself causes potential risks to either the environment (Morris, 2007) or to human health (Malarkey, 2003). Rather the risks are associated with the new phenotypes, and thus, the risks are similar to those with any new crops. GM crops have to pass strict safety assessments before being approved for use in the EU, and therefore, they might be argued to be even safer than conventionally-bred crops (König et al., 2004). As with all technologies, the potential impacts of GM depend on the way the technology is used. When used wisely it may contribute environmental benefits, but if used without care it may result in negative consequences.

Biodiversity conservation should take the landscape context into account. The creation of non-crop semi-natural habitats for wildlife, perhaps in naturally less

productive areas and field margins, may be crucial to maintaining biodiversity within agricultural systems (Feber et al., 2007; Macdonald et al., 2000). In sensitive river catchments, biodiversity conservation areas should be located so that they function as effective buffer zones between agricultural land and water bodies. Agroforestry systems can also enhance the biodiversity and landscapes, offering other benefits in addition, such as carbon sequestration, soil erosion control and reduced nitrogen leaching (Palma et al., 2007). In areas where forests are scarce, biodiversity and landscape benefits can be gained by converting excess agricultural land to sustainable forestry. Converting agricultural land to forests also sequesters carbon and mitigates climate change (Maraseni et al., 2008; Pibumrung et al., 2008).

4.5 Prospects of integrated farming systems

Some studies showed that conventional systems that used best practices for reducing environmental impacts while producing higher yields tend to lead to the lowest environmental impacts often even on an area basis (Basset-Mens and van der Werf, 2005; Korsaeth, 2008; Torstensson et al., 2006). Therefore, the environmental benefits may be optimised by using integrated farming systems that use the best farming technologies for reducing the environmental impacts while producing high yields. Modelling studies by Tuomisto et al. (2012a; 2012b) showed that integrated farming systems that combine the best farming technologies from organic and conventional systems have potential to result in higher environmental performance than either of the systems alone.

5 Conclusions

This meta-analysis has showed that organic farming in Europe has generally lower environmental impacts per unit of area than conventional farming, but due to lower yields and the requirement to build the fertility of land, not always per product unit. The results also showed a wide variation between the impacts within both farming systems. There is not a single organic or conventional farming system, but a range of different systems, and thus, the level of many environmental impacts depend more on farmers' management choices than on the general farming systems.

In our view, there will be no single best farming system for all circumstances. Rather it will be necessary to compose 'optimal' systems from a series of particular practices that relate to specific circumstances, constraints and objectives. Any 'optimal' system will depend on the relative prices that are attached to the alternative outputs and outcomes, whether they are in the form of marketed food, fibre or renewable energy, or whether in the forms of external costs and public goods. These prices will vary geographically and over time. Some systems will continue to be designed to meet the particular standards required for organic production; some will operate in sensitive environments and be required to make special provision for environmental goods, while others will set a higher priority on production. This takes us well beyond the 'organic' versus 'conventional' debate.

Policy needs to recognise and address this complexity and to develop in response to the evolving understanding of the environmental cost-effectiveness of alternative practices and the changing social priorities for environmental systems. Incentives and

norms should be concentrated more on providing incentives for farmers to adopt beneficial practices over damaging practices. Such incentives should also recognise the alternative land use options. In that sense agricultural land should be used efficiently for food production making excess land available for provision of other ecosystem services (e.g. carbon sequestration through sustainable forestry).

Further research is needed to explore the feasibility of assembling the elements of best environmental practices into coherent agricultural systems. These systems then need to be tested through a re-invigorated emphasis on farming systems research that assesses the alternatives not simply in terms of farm level returns but also through comprehensive Life Cycle Assessment of their place within wider food and energy supply chains. Similarly, in coming to an overall view of the relative performance of different systems, it is necessary to attribute relative importance to different types of environmental impacts (Tuomisto et al., 2012b, 2012c). This implies the adoption of some sort of multi-criteria analysis that introduces an implicit or explicit weighting of environmental impacts.

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