Chapter 8
Environmental performance of organic farming

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8.1 Organic farming as a green technology

The environmental impacts of human activities have been increasing with growing populations and industrialization (Meadows et al. 1972). This led to the development of green technologies, which stand out for their positive environmental impacts or the avoided negative environmental impacts. Green technologies feature concepts like sustainability, "cradle-to-cradle" design, input reduction, innovation, and viability.

Organic farming suits this notion of a green technology. Organic farming has emerged in the course of the twentieth century as an environmentally friendly alternative to conventional agriculture (Niggli 2007; Vogt 2007). In the course of rapid structural change, conventional agriculture became increasingly capital intensive, input dependent, and specialized. Bound to strict rules regarding nutrient cycling and input avoidance, organic farming did not follow this path, which led to a significant gap between the two farming systems over time. Lampkin (1990) stresses, however, that several misconceptions exist regarding organic farming: commonly, organic farming is conceived as farming in the pre-1939 style or a production method that does not use chemicals, substitutes mineral fertilizers with organic fertilizers, and bans pesticides. However, the role of agro-ecosystem management and other progressive management practices is often ignored in such conceptions.

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The international umbrella organization of organic agriculture, the International Federation of Organic Agriculture Movements (IFOAM), defines organic agriculture as:

> […] a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved (IFOAM 2009).

This definition highlights the “largely self-sustaining” (Köpke et al. 1997) nature of organic farming as a farming system. Thus, organic farming seems to meet the requirements for being a green technology. According to the above definition, organic farming is sustainable because it does not jeopardize needs of future generations, following the concepts of “cradle-to-cradle” design, input reduction, and sustaining soil fertility. Its worldwide rapid growth of 1.5 million hectares (ha) between 2006 and 2009 (Willer and Kilcher 2009) demonstrates its economic viability and its power to overcome self-imposed system restrictions by innovation.

Certification of organic farms is an important means to both establish credibility for consumers and guarantee a higher willingness-to-pay for organic produce compared to conventional products (Krystallis and Chryssohoidis 2005). Thus, certification generates additional farm income in order to make organic agriculture economically viable. Detailed standards, principles, and aims are set out by IFOAM in the periodically revised “IFOAM Norms.” These contain the “IFOAM Basic Standards” and the “IFOAM Accreditation Criteria” as an international guideline for national standards in organic agriculture. According to Huber et al. (2010) 73 countries have implemented organic legislation, whereas 16 countries are currently in the process of drafting a legislation. In the EU, for instance, Council Regulation (EC) No. 834/2007[1] was also based on the IFOAM Basic Standards and provides a binding framework for EU Member States (IFOAM 2009). More detailed rules for the implementation of organic farming in the member states are set out in Commission Regulation (EC) No. 889/2008.[2] In Switzerland, the federal (country-wide) standards[3] have been developed according to Council Regulation (EEC) No. 2092/91[4] and were updated in 2010 according to Council Regulation (EC) No 834/2007.

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[1] Council Regulation (EC) No. 834/2007 of 28 June 2007 on organic production and labelling of organic products, repealing Regulation (EEC) No. 2092/91, O.J. L 189/21 2007. This regulation was amended by Council Regulation (EC) No. 967/2008 of 29 September 2008, O.J. L 264/1 (2008).

[2] Commission Regulation (EC) No. 889/2008 of 5 September 2008 laying down detailed rules for the implementation of Council Regulation (EC) No. 834/2007 on organic production and labelling of organic products O.J. L 250/1, which was amended by Commission Regulation (EC) No. 1254/2008 of 15 December 2008, amending Regulation (EC) 889/2008 laying down detailed rules for implementation of Council Regulation (EC) No. 834/2007, O.J. L 337/80.

[3] Ordinance on Organic Farming. Verordnung des EVD vom 22. September 1997 über die biologische Landwirtschaft (SR 910.181).

[4] Council Regulation (EEC) No. 2092/91 of 24 June 1991 on organic production of agricultural products and indications referring there to on agricultural products and foodstuffs.

This chapter aims at comparing the environmental impacts of organic agriculture with those of conventional agriculture based on state-of-the-art literature. Furthermore, it aims at discussing methodological implications for the comparison of environmental impacts of farming systems.

Subsequent to this introduction, Sect. 8.2 focuses on the comparison of the environmental impacts of organic farming on biodiversity, climate change, resource depletion, ground and surface water pollution, air quality, and soil fertility with those of conventional farming. Section 8.3 discusses methodological implications of current research on the environmental impacts of farming systems by describing the most important problems of environmental assessments and by suggesting solutions for dealing with these problems. Finally in Sect. 8.4, conclusions are drawn regarding the impacts of organic farming.

8.2 Environmental impacts of organic farming

It is important to assess and quantify how green organic farming actually is. This is done by comparing organic agriculture with other farming systems, usually referred to as "conventional" agriculture. Acknowledging the fact that conventional agricultural practices are already diverse, it is difficult to determine its exact impacts. Moreover, agriculture as it is practiced in reality diverts from laws, standards, and regulations. Thus, there are basically two possible approaches for such a comparison:

- **Normative comparison**, i.e., comparing (minimum) standards: farming systems are assessed according to the environmental standards that have to be fulfilled. For conventional farming this is usually the environmental law or—if existing—cross-compliance standards.
- **Positive comparison**, i.e., comparing real farms: In reality farms divert from standards, laws, and regulations which they have to fulfil. For example, not all farms comply with minimum standards, such as an even nutrient balance. On the other hand other farms voluntarily tend to meet higher standards than they are obliged to.

Either type of comparison as well as blends of both can be found in literature. However, for drawing conclusions in this book chapter (i.e., to assess the environmental performance of organic farming as a green technology in practice), we will opt for the positive comparison.

As shown below, there are many studies identifying the positive and negative environmental effects of organic products or organic agriculture. Environmental impacts are grouped according to the types of natural resources concerned. In order to analyze the environmental impacts of organic farming on biotic and abiotic resources, first, biodiversity and landscape impacts will be looked at. After that climate change mitigation, resource depletion, ground and surface water pollution, air quality, and soil fertility will we reviewed. Social, economic, and ethological impacts of organic agriculture, such as provision of labor in rural regions, health benefits, and an increased animal welfare are not considered in this chapter.

8.2.1 Biodiversity and landscape

Biodiversity can be described according to four different levels. First, diversity within species (genetic diversity) includes the diversity of farm animals and crops. Genetic diversity enables species to adapt to changing environments (e.g., caused by climate change). Second, it is expressed at species level (encompassing faunal and floral diversity), most simply by monitoring the species in selected groups, such as birds or plants in a certain area. Third, biodiversity can be expressed in terms of the regional diversity of habitats and ecosystems in which species live (Christie et al. 2006). Fourth, ecosystem functions describe services delivered by functioning natural systems to humans. One of the key messages of a major study on "The Economics of Ecosystems and Biodiversity" (TEEB) was the "inextricable link between poverty and the loss of ecosystems and biodiversity" (TEEB. 2010).

Owing to agricultural activities, a great variety of ecosystems have been created which, overall, have enhanced biological diversity. On the other hand, agriculture negatively affects biodiversity directly through cultivation practices. Furthermore, it affects biodiversity indirectly through nitrogen emissions into the air and CO_2 emissions into the atmosphere. On land under intensive agricultural cultivation, biodiversity decreases significantly because of the high nutrient influx, high cutting frequencies on meadows, high stocking rates, use of pesticides, and modern methods of processing cut grass (Knop et al. 2006). In Alpine lowlands many diverse agricultural ecosystems have disappeared, while in the mountain regions two parallel trends are apparent: the intensification of productive areas and the abandonment of unproductive but ecologically valuable areas (Aeschenbacher and Badertscher 2008).

Biodiversity effects are among the most frequently studied environmental impacts of organic agriculture. Recent metastudies (Bengtsson et al. 2005; Fuller et al. 2005; Hole et al. 2005) show clear differences between organic and conventional farming systems. In very rare cases, organic production was found to have negative impacts, although this was outweighed by studies showing positive impacts. The differences vary among taxonomic groups, but for each species group large differences were found (Table 8.1). On average, about 50% greater species diversity was achieved on organic farms (Niggli et al. 2008).

Organic farming practices are most beneficial for birds, predatory insects, spiders, soil organisms, and the arable weed flora, while pests and indifferent organisms do not show different levels of abundance in the farming systems. Furthermore, differences in arable land between the farming systems are more pronounced than on grassland (Niggli et al. 2008).

Apart from differences at species-group level, structural differences at farm level are prevalent between organic and nonorganic farms (Gibson et al. 2007; Schader et al. 2008b). In addition, Boutin et al. (2008) identified higher species richness in seminatural habitats on organic farms compared with conventional farms.

Genetic biodiversity is influenced both positively and negatively by organic farming. On the one hand many organic farmers cultivate rare plant and animal species on their farms (e.g., because they are better adapted to local conditions); on the

Table 8.1 Number of studies analyzing the impacts of organic farming on biodiversity with respect to various *taxa* on the basis of 76 comparative studies[a]

	Impacts of organic farming		
Taxa	Positive	No difference	Negative
Plants	16	2	0
Birds	11	2	0
Mammals	3	0	0
Arthropods			
Beetles	15	4	5
Spiders	9	4	0
Butterflies	2	1	0
Bees	2	0	0
Other arthropods[b]	8	3	1
Bacteria, fungi and nematodes	12	8	0
Earthworms	8	4	2
Total	**87**	**28**	**8**

[a]Updated using studies from 2004 to 2008
[b]Mites, bugs, millipedes, flies, and wasps
Source: Hole et al. (2005), updated by Niggli et al. (2008)

other hand the restriction on admission of varieties hampers genetic diversity. Because there is insufficient scientific evidence on the impacts of organic farming on genetic biodiversity, we assume both farming systems to perform equally well.

Concerning **landscape** and habitat diversity, organic farming may perform better because of more diverse crop rotations (Norton et al. 2009) and higher implementation rates of structural elements such as hedges and fruit trees (Schader et al. 2009). However, landscape effects are very farm and site specific. Therefore, no general trend can be determined (Steiner 2006).

In a Swiss study, Schader (2009) analyzed the average habitat quality by combining an economic sector-representative farm group model (Sanders et al. 2005) with a species diversity model by Jeanneret et al. (2006) for different farm types, regions, and all of Switzerland. Figure 8.1 shows the average habitat quality in different farming systems for five species groups (that are on the red list of endangered species) and 11 indicator species as well as for an overall habitat quality indicator. Average habitat quality is expressed as a percentage of a hypothetical maximum habitat quality, which is achieved in an optimal habitat quality under optimal farm management conditions (=100%). The study showed that the average habitat quality over all species is 25% on organic farms and 16% on conventional farms. Schader attributed the 55% improvement of habitat quality to the on-average higher share of grassland, the on-average lower grassland intensity (lower stocking density, lower fertilization, fewer cuts) and the different farm type and regional distribution.

In summary, studies attribute the higher biodiversity in organic systems to the following factors: (a) ban on herbicides and artificial pesticides, (b) ban on mineral fertilizers, (c) more diverse rotations, (d) lower organic fertilization, (e) careful tillage, (f) a higher share of seminatural habitats in total UAA (Utilized Agricultural Area) (Bengtsson et al. 2005; Fuller et al. 2005; Hole et al. 2005).

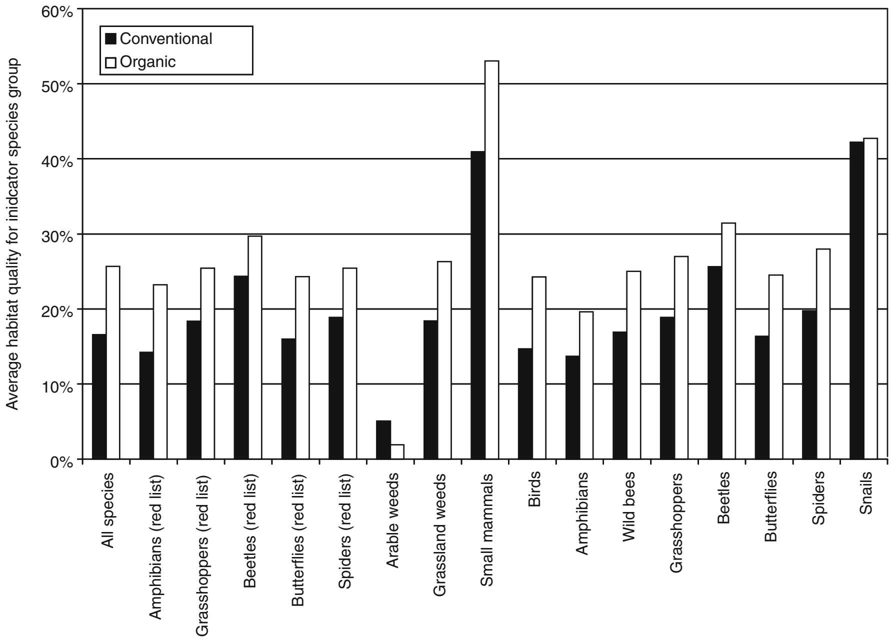

Fig. 8.1 Average habitat quality by species group (average over all regions and farm types, 2006/2007)

8.2.2 *Resource depletion*

Resource depletion is a problem of similar magnitude against the background of industrial civilizations' growing dependency on fossil fuels (Meadows et al. 1972). The debate about peak oil (i.e., the point in time when the maximum rate of global oil extraction has been reached) is currently increasing in intensity (Zittel and Schindler 2007). Agriculture, once a net energy producer has today become a net energy consumer for some commodities. Given the need for efficient resource use, energy use has become a standard environmental indicator (Pimentel et al. 2005; Frischknecht et al. 2007).

Nevertheless, many studies suggest that the whole food system (agricultural production, food processing, packaging, and distribution) makes up a large percentage of energy consumption (Ziesemer 2007). Studies comparing crop and animal products conclude that crop products have much higher energy efficiency per unit of digestible energy than produce from animal production (Pimentel and Pimentel 2005).

The impacts of organic agriculture on energy use can be analyzed on the basis of different functional units (Halberg 2008). While some studies use "area" as a unit (Haas et al. 2001), others take the weight of output from the farming system as a reference (Grönroos et al. 2006). Although the latter approach is in line with the standard procedure within life cycle assessments (Heijungs et al. 1992) and illustrates energy use per unit of food produced, it still has weaknesses when it comes to analyzing agricultural systems. Often, research on farming systems encompasses consideration of multiple outputs. Either these outputs need to be expressed in a single unit, or an allocation of the energy use has to be performed, or again, byproducts need to be deducted to enable a comparison across all products (Schader et al. 2008a). A product-related assessment additionally involves the determination of the functional unit. However, the scorings related to weight, volume, calories or protein might produce highly varied results.

Stolze et al. (2000) also concluded that organic farming systems perform better than conventional ones in terms of energy use. The energy use of growing permanent crops (olive, citrus, vineyards) and arable crops (grains, pulses, etc.) is, related to product output and area cultivated, lower in organic than in conventional farming systems. However, growing potatoes organically can require equal or more energy per product output than doing so conventionally. Also Lampkin's review (2007) identified that most product- and area-related energy use assessments of organic farming to date show a lower energy use per-ha. Due to the generally lower productivity of organic farming, per-ha comparisons reveal higher differences than product-based comparisons. Thus the choice of the appropriate functional unit is crucial when comparing organic and conventional agriculture (see Sect. 8.3.2.1 in the discussion section of this chapter). Haas et al. (2001) compared organic and conventional grassland farms in southern Germany. They found a 44–46% lower energy use per ha and per ton of milk. Thomassen et al. (2008b) also analyzed milk production and found that the energy efficiency of organic production was significantly higher compared to conventional production. They concluded that the use of concentrate feed in particular is a major driver and has potential for reducing energy use.

Grönroos et al. (2006) calculated that fossil energy use for organic rye bread and milk was lower by 13% for rye bread and 31% for milk—compared with conventional products. In a cradle-to-(farm-) gate perspective, the difference is even higher, with organic products consuming only 50% of the energy use of conventional products. Similar results were generated by Hoeppner et al. (2005) who compared the energy use throughout a rotation. Energy use and energy output were 50% and 30% lower, respectively, over organic rotations in a long-term field experiment, which results in higher energy efficiency (energy use per product) of organic farming compared with conventional farming.

Nemecek et al. (2005) demonstrated a lower energy use per ha and per product unit overall in organic systems for all major crops in Switzerland. This was done by analyzing data from long-term field experiments and generating subsequent calculations aimed at generalizing the results for Switzerland. An exception to this is potatoes, where a slightly higher energy use was calculated per ton of organic potatoes.

Figure 8.2 shows the differences in energy use per ha between organic and conventional farm types in Switzerland, based on a representative farm sample (Schader 2009). Pig and poultry farms have been excluded from the graph in order to maintain its readability for the other farm groups; because of the high stocking rates and

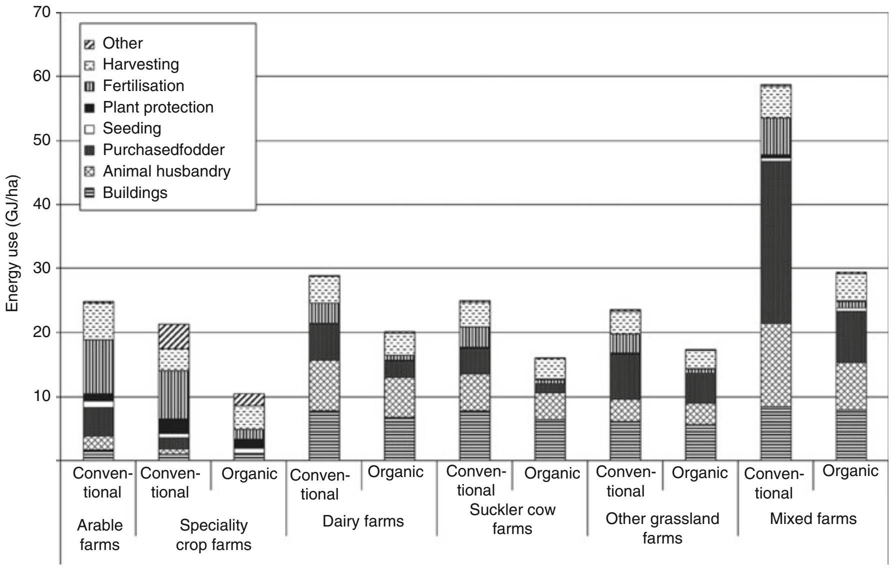

Fig. 8.2 Fossil energy use per ha on conventional and organic farms by farm type (2006/2007)

the high share of cereal-fed livestock, these farms have a calculative energy demand of 195 GJ/ha (not shown in Fig. 8.2). Beside pig and poultry farms, conventional mixed farms have the highest total energy use (60 GJ/ha). The average energy use (as a sum of all energy use components) of dairy, suckler cow, other grassland, arable and finally speciality crop farms ranges from 20 to 30 GJ/ha. Organic counterparts have an energy use which is about a third lower (10–20 GJ/ha), except for mixed farms where the average reduction in energy use amounts to 50%. Schader (2009) attributes the lower energy use of organic farms to lower quantities of purchased feedstuffs, particularly concentrates, lower stocking densities, the ban of mineral nitrogen fertilizers, and the absence of highly intensified specialized pig and poultry farms.

In summary, organic farming has a lower energy use per ha, and in most cases also per mass of product, than conventional farming. There are only a few exceptions on the crop-production side, notably potatoes, with organic systems displaying lower energy efficiency owing to low relative productivity levels. While milk production is more efficient on organic farms, poultry production has shown slightly lower energy efficiency. Thus, the quantitative advantage of organic farming depends crucially on the product, the geographic region, and the assumptions of the study.

Besides depletion of energy resources, in particular phosphorus (P) resources are exploited as phosphorus is applied on agriculture fields in large quantities. Because of the ban of easily soluble phosphate and potassium fertilizers and the widely closed nutrient cycles, organic farming leads to a lower depletion of nutrient resources. Positive impacts on P resource depletion are backed by life-cycle assessments (Nemecek et al. 2005).

There is little evidence on the impact of organic farming on water use efficiency (Stolze et al. 2000); however, because of lower yields organic farms might tend to use more water per unit of output. On the other hand, lower biomass production per area tends to make organic agriculture particularly attractive for areas with water scarcity. Furthermore, as animal production is most water intensive, the lower stocking rates of organic farms might lead to lower water use per ha.

8.2.3 Climate change

Climate change has been perceived for decades as a significant global environmental problem. Over the last years in particular environmental awareness has increased markedly within the general population, partly because of reporting on the international negotiations in relation to the Kyoto Protocol, and partly because of more visible impacts of climate change on ecosystems such as glaciers or Polar Regions. The higher incidence of natural disasters such as hurricanes, droughts, and floods has also contributed to the growing awareness of climate-change over the last few years (IPCC 2007). Recent studies estimate that the cost of current and projected levels of greenhouse gas emissions and the climate change caused by them exceeds their abatement costs (Stern 2007). Because agricultural production has an impact on all the three major greenhouse gases (CO_2, CH_4, and N_2O), it is perceived as a crucial and potentially cost-effective lever for mitigating climate change (Smith et al. 2007).

About 12–14% of global greenhouse gas emissions stem from the agricultural sector (Smith et al. 2007). However, while CH_4 and N_2O emissions are predominantly attributable to agriculture, the share of the agricultural sector in terms of CO_2 emissions caused mainly by burning fossil fuel is disproportionately small.

The literature review of Stolze et al. (2000) found that organic agriculture bears a lower CO_2 and NH_3 gas emission potential than conventional farming systems. However, because of lack of scientific literature, they concluded that there were no differences between these farming systems with respect to N_2O and CH_4.

As is the case for energy use, the impacts of organic agriculture on CO_2 emissions can be analyzed on the basis of different functional units (Halberg 2008). While some studies use "area" as a unit (Haas et al. 2001), most studies take the weight of output from the farming system as a reference. Since the performance of organic agriculture regarding CO_2 emissions is highly correlated to energy use, the same arguments apply as for the discussion of energy use in the above section. Unlike that of energy use though, net emissions of CO_2 (i.e., gross emissions subtracted by the sequestration rate) need to be taken into account. There are indications of better performance regarding carbon sequestration (Olesen et al. 2006; Niggli et al. 2009). Several long-term trials from the United States, Germany, and Switzerland (Mäder et al. 2002) show that organic farming systems are able to sequester on average 590 kg/ha per year more carbon from the atmosphere than the best performing conventional counterparts.

Table 8.2 Comparison of N_2O emissions per unit of area under conventional and organic management

References	Country	Method	Emissions in organic systems		
			Lower	Equal	Higher
Petersen et al. (2006)	Austria, Denmark, Finland, Italy, UK	Field measurement	x		
Chirinda et al. (2010)	Denmark	Field measurement		x	
Küstermann et al. (2008)	Germany	Modelling	x		
Flessa et al. (2002)	Germany	Field measurement	x[a]		
Sehy (2004)	Germany	Field measurement	x[a]		
Lynch (2008)	Canada	Field measurement	x		
Nemecek et al. (2005)	Switzerland	Modelling	x[b]		
Hansen et al. (2008)	Norway	Field measurement	x		

'X' indicates scientific evidence on higher, equal, or lower N_2O emissions under organic management
[a]No difference in output-related emissions
[b]Output related emissions are lower in organic systems
Source: Gattinger et al. (2010), Adapted

In summary, organic farming has lower CO_2 emissions per ha and, in most cases, also per ton of product than conventional farming. There are only a few exceptions on the crop-production side, notably potatoes, with organic systems displaying lower energy efficiency due to low relative productivity levels. Although milk production is more efficient on organic farms, poultry production has shown slightly lower energy efficiency. Thus, the quantitative advantage of organic farming depends crucially on the product, the geographic region, and the assumptions of the study. Furthermore, organic farming has a potential to mitigate climate change by reducing greenhouse gas emissions through their sequestration in soil.

About 75% of CH_4 emissions stem from enteric fermentation of ruminants. There are two different perspectives on the impact of organic farming on CH_4 emissions per unit of output. On the one hand, CH_4 emissions could be higher owing to less output (milk or meat) per livestock unit and time. On the other hand, many organic farms tend to keep dairy cows for more lactation periods than conventional farms, which lower the CH_4 emissions per unit of output during the growing up phase. When looking at the emissions per hectare (e.g., from an agri-environmental policy evaluator's perspective), organic farming has lower CH_4 emissions because of lower stocking densities.

So far, there are only a few studies available that compare N_2O-emissions from organic and conventional farming systems (Table 8.2). Chirinda et al. (2010) found no differences in N_2O-emissions between farming systems, while all other authors found lower N_2O-emissions per ha. Calculated per output quantity, N_2O-emissions in organic systems where equal to non-organic farming systems according to Flessa et al. (2002) and Sehy (2004) while Nemecek et al. (2005) modelled 18% lower N_2O-emissions in organic farming systems than in conventional ones.

Because in general, there is a linear relationship between N-Input und N_2O-release and in organic farming systems N-supply is up to 50% lower than in conventional farming systems, Gattinger et al. (2010) conclude that organic farming systems have a lower N_2O-emission potential than conventional farming systems.

In summary, data uncertainty concerning N_2O emissions from different fertilizers and from the soil does not allow general conclusions to be drawn on the impact of organic farming. N_2O is emitted from agricultural soils at specific periods of time, depending on nitrogen, carbon, and oxygen in the soil. Influencing factors on N_2O emissions are type and amount of nitrogen fertilization and water logging. In general, however, N_2O emissions could be lower owing to lower nitrogen fertilization rates and the applications of fertilizers with lower nitrogen concentration. On the other hand, N_2O emissions could be higher in organic systems per unit of output due to the higher land use.

8.2.4 Ground and surface water pollution

Eutrophication is defined as nutrient enrichment in sensitive ecosystems (UNECE 1999). Eutrophication entails various environmental impacts that cause both the loss of biodiversity and negative impacts on human health. Eutrophication leads to excessive growth of algae and excessive oxygen demand, with anaerobic conditions leading to foul smelling surface waters and fish death. These effects of eutrophication can be understood as societal costs, either in terms of abatement, purification, or restoration costs, or as damage costs if the negative impacts of eutrophication are not abated or fixed. Besides eutrophication, influxes of toxic substances particularly into surface water can pose a significant environmental threat because it can lead to severe harm to aquatic life.

The main environmental risks entailed by agricultural production in relation to ground and surface water pollution involve eutrophication with nitrogen and phosphorus and pesticide emissions. The leaching of mobile nitrates into ground and surface water and gaseous emissions such as ammonia (NH_3) from organic fertilizers are the major contributors to nitrogen eutrophication. Ammonia emissions affect ecosystems like forests, swamps and diverse meadows, which require low nitrogen levels. Furthermore, ammonia emissions into ecosystems cause acidification and the release of toxic substances including heavy metals.

Nitrate pollution in the lowlands has been the most severe environmental problem resulting from post-war policies (Gruber 1992). These policies provided incentives to run intensive, highly-yielding agricultural production involving heavy nutrient surpluses.

Phosphorus is relatively immobile in soils but can be emitted from agricultural systems to surface waters by erosion and run-off processes. While phosphorus rarely represents an environmental problem in rivers, it causes algae growth in lakes and seas, because normally phosphorus is the limiting nutrient for algae growth.

The decomposition of this additional plant material reduces the amount of oxygen. Finally, fauna die because of anaerobic conditions. Phosphorus emissions from agriculture give rise to high societal costs because of bad odors, costs of treatment, and the hindrance of recreational activities.

The reduction of nitrogen and phosphorus eutrophication demands efficient use of these nutrients (Herzog and Richner 2005). Evaluations have identified that the problem of nitrate leaching occurs predominantly in arable farming systems, although leaching can also occur from grassland receiving high fertilizer inputs. Therefore, Herzog and Richner (2005) suggest that farms should no longer be permitted to have a 10% nutrient surplus. Apart from systems that rely heavily on imported manures (e.g., horticultural systems), there is no nutrient surplus in organic systems, because nutrient import onto the farm is restricted for both feedstuffs and mineral fertilizer.

Several studies show that nitrogen leaching can be reduced by 40–64% through organic farming (e.g., Edwards et al. 1990; Younie and Watson 1992; Eltun 1995; Condron et al. 2000; Goulding 2000; Haas et al. 2001; Kirchmann and Bergström 2001; Mäder et al. 2002; Stopes et al. 2002; Auerswald et al. 2003; Pacini et al. 2003; Shepherd et al. 2003; Osterburg and Runge 2007).

In contrast, Nemecek et al. (2005) found higher eutrophication impacts for some organic crops compared to their conventional counterparts per kg output. In some places, these higher nutrient loads on arable land are attributed to the greater use of organic fertilizers in the organic system, because the life cycle assessments used by Nemecek et al. (2005) assume relatively high fertilization rates for organic farms.

Taking the data by Nemecek et al. (2005) and projecting them at sector level using statistical data and an economic model, Schader (2009) found on average 35% lower eutrophication rates. Figure 8.3 shows the average eutrophication with nitrogen for average conventional and organic farm types in Switzerland. As can be seen, nitrate eutrophication rates on different farm types vary drastically. Whereas farms specialized in animal production usually have nitrate leaching rates below 20 kg N-eq/ha, organic counterparts show a more than 50% reduction. Mixed farms have a higher nitrate leaching rate because of a higher share of arable land, whereas organic farms have a 20% lower eutrophication rate than their conventional counterparts. According to Schader (2009), there are only a very few organic specialized plant production farms, which makes it difficult to model nitrate leaching at sector level.

In the same study phosphorus eutrophication was also modelled, which showed a 10–20% decrease of phosphorus eutrophication on organic farm types, compared with their conventional counterparts (Fig. 8.4). Other studies on impacts of organic farming on eutrophication with phosphorus are scarce. However, acknowledging the fact that literature indicates significant efforts of organic farming to improve soil quality and to reduce erosion risk (see section on soil fertility below), phosphorus runoff can be assumed to be lower in organic systems (Shepherd et al. 2003; Schader 2009).

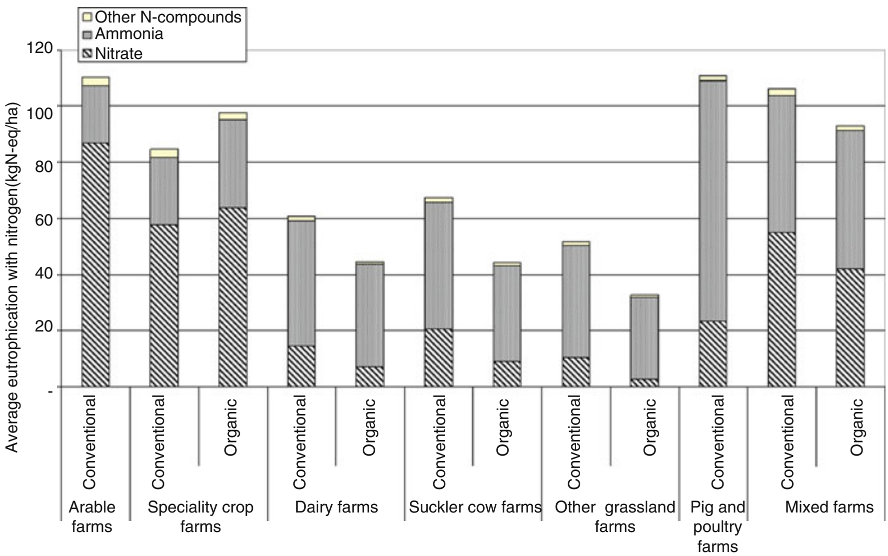

Fig. 8.3 Nitrogen eutrophication per ha on conventional and organic farms by farm type (2006/2007)

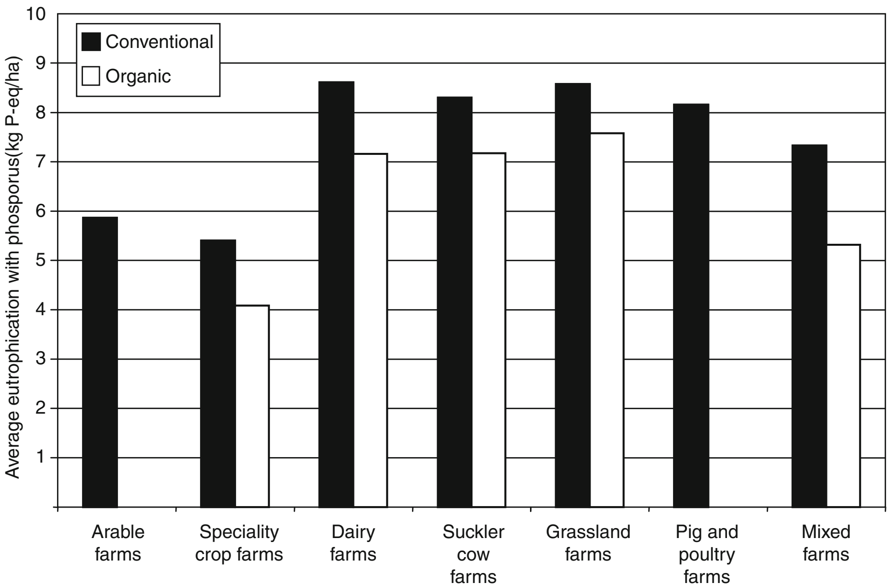

Fig. 8.4 Phosphorus eutrophication on conventional and organic farms by farm type (2006/2007)

Summing up, there are three facts underlining a lower eutrophication potential of organic farming:

- Organic farming systems have lower nutrient levels, which reduces the absolute quantity of nutrient loads that can be emitted from the system due to lower stocking rates and the ban of mineral nitrogen fertilizers.
- The quantity of directly available nitrogen is much lower in organically managed soils.
- Because nutrients cannot be imported easily into the systems, the opportunity cost (shadow price) of nitrogen losses is higher for organic farms than for conventional farms (Stolze et al. 2000). This implies a need for more efficient nutrient management in organic systems, although this does not eliminate losses. In addition, nitrate leaching can be high at the point of transition from the fertility building phase of the rotation to the cropping phase.

8.2.5 Air quality

The main environmental risks in terms of air pollution are entailed by agricultural production gaseous emissions such as ammonia (NH_3) from organic fertilizers and gaseous emissions from pesticides. Furthermore, ammonia emissions affect ecosystems like forests, swamps, and diverse meadows, which require low nitrogen levels.

There are only few comparisons between ammonia emissions from organic systems compared with conventional systems. In a study on representative life cycle assessment (LCA) approach by Schader (2009), ammonia emissions were found to be almost equal in both systems for each farm type. The fact, however, that specialized pig and poultry farms are hardly feasible according to organic standards, leads to a substantial reduction of stocking densities and thus to a reduction of ammonia emissions for the whole sector. Nevertheless the study assumed that fertilization rates were equal to the nutrient needs of the crops. In reality however, in organic systems nitrogen is applied in lower amounts, which also leads to reductions in ammonia emissions. Finally, opportunity cost of each kg of ammonia lost from the system is much higher on organic farms, because nitrogen cannot be fed into the system easily (Stolze et al. 2000). Therefore, organic farms should have a higher incentive to implement strategies for reducing the loss of nitrogen via ammonia emissions from the system.

Thus, per unit area, ammonia emissions are lower in organic systems because of lower stocking densities, whereas per livestock unit or milk/meat produced, emissions may be equal if productivity is in a similar range. Because most pesticides are banned in organic systems, there are substantially lower pesticide emissions from organic systems both per unit area and production unit.

8.2.6 Soil fertility

Soil is one of the major production factors of agriculture and therefore an essential natural resource for ensuring food security. Yet agricultural production at the same time poses a threat to this resource. According to Lal (2004) agricultural soils have lost a great share of their organic matter to wind and water erosion because of intensive agriculture, which is responsible for the loss of one third of fertile lands from 1955 to 1995 (Pimentel et al. 1995).

Organic agriculture encompasses a number of different activities within its system approach, which aim at increasing organic matter content in the soil. Most important is the ban of mineral fertilizers, which necessitates meeting the nutrient needs of the crops by organic fertilizers (Mäder et al. 2002). Furthermore, the fundamental importance of a crop rotation including short-term ley (i.e., nonpermanent meadows) supports the development of fertile soils (Pimentel et al. 2005).

These activities also favor biological activity in the soil. As has been reported above in the section on biodiversity, there is clear scientific evidence that soils under organic management have higher biological activity both in terms of species and general biomass (Mäder et al. 2002; Pfiffner and Luka 2007). Furthermore, organic farmers need to pay attention to soil fertility in the long term, because the ability of the soil to capture nutrients is crucial in organic systems (Köpke 2003).

Soil erosion has already been discussed in the context of phosphorus runoff, because erosion processes are the dominant driver for phosphorus losses. There are some studies reporting a beneficial effect of organic farming on soil erosion and soil structure (Siegrist et al. 1998; Stockdale et al. 2001; Shepherd et al. 2002). Organic farming performs much better in terms of soil biological activity than nonorganic farming. Soil erosion and organic matter content are also affected positively by organic practices.

8.3 Methodological implications for a comparison of farming systems

In this section we first present a methodological classification of studies reported according to nine characteristics. Second, we discuss implications for methodological choices when comparing organic and conventional production systems.

8.3.1 Classification of methodological characteristics

When comparing the environmental performance of organic farming with that of conventional farming, there are nine different methodological characteristics by which studies can be classified following the four main phases of life cycle assessment (Table 8.3). Five of these refer primarily to the phases "goal and scope definition" and

Table 8.3 Classification of methodological choices for comparing the environmental impacts of organic and conventional farming

LCA phase	Characteristic	Possible methodological choices
Goal and scope definition and life cycle inventory	Scope	Field, farm, region, sector-wide, global
	Functional unit	Area, mass, energy, protein, net value added
	Stage of processing	Raw product, processed and packed products and supply chains
	Life cycle perspective	Life-cycle perspective (cradle-to-grave, cradle-to-gate), economic perspective
	Normativeness	Based on statistics (positive view), based on normative models
Life cycle impact assessment and interpretation phase	Time perspective	Static view, dynamic view
	Scope of impacts	Single impact category, several impacts, only environmental impacts, all impacts relevant for sustainability of farming system
	View on sustainability	Weak sustainability, strong sustainability
	Units of impacts	Physical/biological units, relative effects, monetary units

"life cycle inventory," whereas the remaining four focus more on the phases "life cycle impact assessment" and "interpretation" of results.

First, the scope of the study can range from single fields, products, rotations, and farms to representative studies for whole sectors or even studies at the global level, although most studies are conducted at field or product level.

Second, the functional unit may either be related to area or production. Production can be measured by several units such as mass, energy content, protein content, or net value added.

Third, both raw products and processed products or entire supply chains can be assessed. Although standards for organic farming increasingly cover standards on processing (e.g., choice of additives), only raw agricultural products were assessed in the survey and in the bulk of studies found.

Fourth, studies can take on either a life cycle perspective or an economic perspective. Whereas the economic perspective usually covers only the resources used directly in the sector, a life cycle perspective covers also environmental impacts of inputs that occurred in earlier stages, e.g., fossil energy use for producing mineral nitrogen fertilizers (cradle-to-gate perspective). Even use and disposal phase can be covered, e.g., for specific inputs, buildings, or packaging materials (cradle-to-gate perspective). Commonly the impact categories climate change, resource depletion water and air pollution are assessed by agricultural life cycle assessments, usually based on the ISO Norms 14040 and 14044. Because of methodological difficulties, biodiversity and soil quality are only rarely assessed using LCAs.

Fifth, as already described in Sect. 8.2, the degree of normativeness of assumptions is different across studies. For example, a normative approach to assessing fertilization would be to assume that nutrient needs of plants are covered, whereas a positive approach is to calculate impacts based on the fertilizer amounts actually used. Most studies have a significant degree of normative assumptions in the models they use, due to a lack of empirical data.

Sixth, studies cover a static or dynamic perspective. All of the LCA studies followed a static approach, as the development of dynamic LCA models is still in its infancy.

Seventh, studies either analyze only a single environmental category, several, or they try to cover all relevant categories. Furthermore, social and economic impact categories are taken into consideration if aimng at a full coverage of sustainability.

Eighth, environmental impacts can either be aggregated to a single score or remain as several individual indicators. Whereas the first approach is in accordance with the notion of weak sustainability (i.e., full substitution of natural resources), the latter would be a strong sustainability approach.

Ninth, impacts can be expressed either in physical or monetary terms. Physical terms are used most frequently, whereas monetary terms are mostly used in economic comparisons. The latter approach is particularly useful if societal or farm-level costs and the environmental performance need to be compared with each other (e.g., using cost-benefit analysis).

8.3.2 Discussion of methodological implications

The large number of different methodological characteristics illustrates that finding appropriate methods for comparing farming systems is more difficult than for many other goods and services. Depending on the goal of the study and the geographic context, methodological choices can be very different (Hospido et al. 2010).

Life cycle assessments are a generally useful approach for assessing farming systems. Significant methodological advances in the assessment of environmental impacts of agriculture and food products were realized, owing to: (a) the introduction of life cycle thinking, which explicitly includes off-farm environmental impacts (e.g., owing to the production of fertilizers imported by the farm); and (b) the consideration of several environmental impact categories at once, instead of concentrating on a single impact category.

Yet, we see nine methodological problems with some of the current environmental impact assessments, particularly with LCAs, of farming systems. Building on the work by Reap et al. (2008a, b), we describe them and propose ways to address them in practical LCA work.

8.3.2.1 Consideration of the multifunctional character of agriculture

During the last decades the role of agriculture shifted from its mere production function to a multifunctional role. Multifunctionality acknowledges the fact that agriculture fulfils multiple roles in society. Apart from producing food and fibers, agriculture is carried out for landscape maintenance, conservation of natural resources, and cultural purposes (OECD 2001).

The functional unit needs to be chosen according to the function of agriculture which is addressed. Within a multifunctional setting several functions should be assessed. For addressing the function of generation of rural incomes, it makes sense to use "net value added" as a functional unit. Furthermore, in the case of agri-environmental policy evaluations (e.g., for direct payments), particularly agriculture's role to cultivate land is addressed. In such a case "area" is the most appropriate functional unit to consider.

If agriculture or farming systems as such are evaluated, it is necessary to consider all relevant functional units. Thus, the functional unit "area" can be seen as addressing the role of agriculture to maintain landscape, "digestible energy" addresses the function of providing food to the population, and finally "farm income" or "net value added" addresses the function of agriculture to provide rural livelihoods. However, multiple functional units are rarely considered, because this complicates both the research and the communication of the results. Furthermore, current ISO Standards do not offer sufficient guidance on multiple functional units.

8.3.2.2 Covering land use impacts

Land use impacts, for example, the change of biodiversity on agricultural land or soil fertility, can be related to an area more plausibly than to production (Schader et al. 2008c). This leads to the fact that many life cycle assessments neglect or even ignore these impact categories, although the categories are key indicators for comparing both farming systems. We argue that every full life cycle assessment should address biodiversity and soil fertility impacts if a comparison between organic and conventional agriculture is made and general conclusions on the systems' performance are drawn.

8.3.2.3 Heterogeneity of products

A mass-related functional unit is often used to visualize the environmental performance of a specific food product to the consumer. Food quality, however, is not considered appropriately in this functional unit as we speak of "heterogeneous products" when comparing conventional and organic products. The heterogeneity can be illustrated using the higher willingness-to-pay of consumers for organic produce (Krystallis and Chryssohoidis 2005), because the organic product may fulfil further functions and uses. As we know from many consumer studies, at least in Europe and North America, the choice of food products is often made irrespective of energy content. Often products are consumed even because of their low energy content.

Furthermore, many consumers are willing to pay more for organic products not only because of the environmental benefits but for expected health benefits. Although not a universally held opinion, there are indications that product quality of organic products is higher, particularly because of a lower risk of pesticide contamination and higher concentrations of nutritionally desirable fatty acids and antioxidants (Woese et al. 1997; Butler et al. 2008). We suggest that this higher product quality needs to be taken into account by choosing a monetary functional unit (e.g., consumer price).

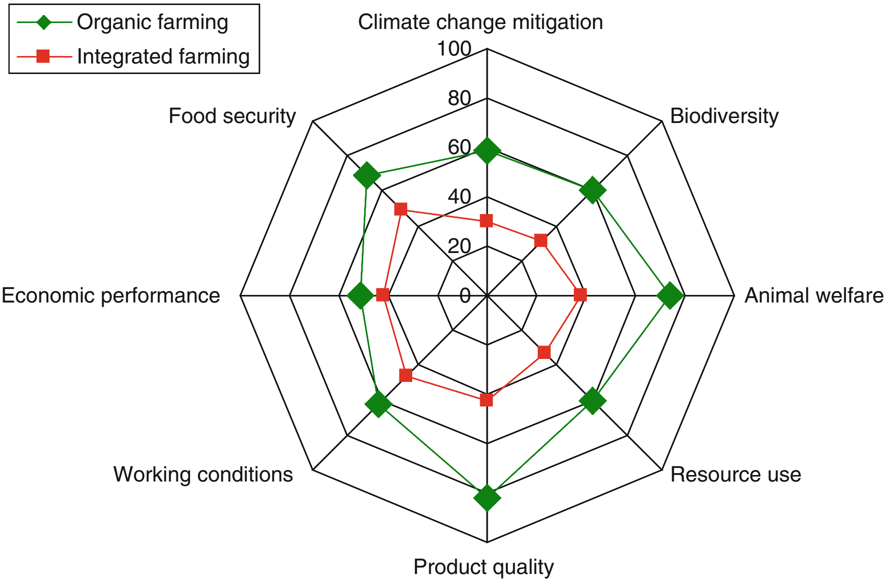

Fig. 8.5 Sustainability assessment of Swiss organic agriculture

8.3.2.4 Covering social and economic aspects of sustainable development

With the rise of the notion of sustainability (i.e., contribution to sustainable development) more comprehensive assessments of organic farming, including economic and social indicators, are demanded. The great complexity of a full coverage of sustainability, however, requires neglecting many details that would be taken into account in studies addressing specific environmental categories. Figure 8.5 shows results from a sustainability assessment using the nominal group technique as a participatory approach (Delbecq et al. 1975; Jeffreys 2002). The results show that neither organic nor conventional/integrated farming can be characterized as fully sustainable (=100%). Whereas conventional farming reaches sustainability scoring of 30–50%, organic agriculture scores better in all categories that were assessed. In particular regarding product quality, animal welfare, and biodiversity, organic agriculture was evaluated with nearly 80% sustainability scorings. Around 60% of sustainability score was given for the environmental indicators biodiversity, climate change mitigation, and resource use (Schader and Stolze 2010).

If a credible judgement on organic farming is to be made, all relevant impacts on sustainability need to be analyzed and made transparent. Whether approaches such as “social LCA” and “life cycle costing” are appropriate and feasible ways for this remains to be determined. It needs to be stressed, however, that not all impacts that cannot be quantitatively assessed are unimportant. We therefore suggest including qualitative assessments if quantitative data is weak. Furthermore, inter and transdisciplinary research needs to be reinforced in order to combine LCA methodologies with social, economic and psychological methods.

8.3.2.5 Regional variations in natural system capacity and environmental legislation

Both natural system capacity and environmental legislation varies regionally and by country. Therefore, it matters for some environmental impacts where the pollution occurs. Attributional LCAs tend to neglect this aspect by using "production mass" as a sole functional unit. This problem has been addressed in water footprinting methods, but is also relevant for eutrophication impacts, as the following example will illustrate.

A system inherent feature of organic agriculture is a lower intensity of production. Particularly stocking rates are restricted but also other restrictions apply, depending on the geographical context. These restrictions, on the one hand lead to lower environmental impacts per ha and lead to lower productivity per area, at least in developed countries (Badgley et al. 2006). If a policy aim is to reduce the eutrophication of groundwater in a particular region, policy-makers might try to reduce the amount of eutrophication in the region by lowering intensity of agriculture (e.g., by supporting the conversion of farmers to organic agriculture). The resulting production loss/gap might be compensated by moving production to regions with lower production cost (a) because of better site conditions or (b) because of less strict environmental legislation, causing indirect land use changes.

Thus, assessing impacts that are purely production-related can lead to wrong conclusions (i.e., a recommendation to intensify grain production if the gains in production mass outweigh the higher eutrophication). We therefore argue that production-related LCAs need to take into account indirect land use changes via consequential approaches, the capacity of natural systems, and define region-specific maximally allowed pollution rates, at least if applied at a larger scale or if results from a case study are generalized.

8.3.2.6 Consideration of the whole farm

A generic thought behind organic farming is the cradle-to-cradle design. Plant production activities provide animal feed and animal production activities provide organic fertilizer. One without the other is impossible and will therefore lead to flawed assessment results. Thus, there are several systems closely interlinked with each other on a mixed farm. Up to which point should emissions from organic fertilizers be allocated towards animal production and plant production? A consistent approach needs to be found, since results of different ways of allocation or system expansion can lead to very different results (Cederberg and Stadig 2003). If a whole farm approach is used, we suggest emissions from organic manure are attributed to animal production on the basis of the principle of economic allocation.

8.3.2.7 Normative assumptions

As was discussed above, the normativeness of assumptions creates a bias of results for the benefit for intensification of agriculture. For example, the higher the fertilizer amounts and the more toxic the pesticides are, the higher the risk of negative environmental impacts if substances are applied incorrectly. If risks of an improper use of the substances are not considered and good agricultural practice is assumed, a bias towards intensive agriculture is generated. We suggest taking into account the risks of toxic substances owing to improper use and over fertilization. Furthermore, empirical data need to be used as much as possible for rendering normative assumptions superfluous.

8.3.2.8 Resulting decisions of agents are not taken into account

There has been a long debate about attributional and consequential LCAs during the last years. With few exceptions, consequential LCAs are hardly ever applied in the agricultural field (Schmidt 2008; Thomassen et al. 2008a). However, the consequential perspective seems to be important in particular in agricultural LCAs. As indicated by the standards and restrictions of organic production, the management of organic farming systems differs systematically from that of nonorganic farming systems. Furthermore, agri-environmental schemes usually provide farmers low symbolic capital, which thus provides no incentive to change farmer's long-term behavior with respect to environmentally friendly farm management practices. Because organic systems are based more on the "naturalness" of the production, organic farmers might have a different system of cultural capital generation (Burton et al. 2008). Studies suggest that applying nature conservation measures is of higher symbolic capital for organic farmers (Stotten 2008), which might result in differences in uptake of agri-environmental schemes (Schader et al. 2008b). These differences in attitudes between organic and conventional farmers imply multiple and systematic effects on the environment (Morris et al. 2001; OECD 2004), which are disregarded in most of the LCA studies that were reviewed. Thus, we argue that if organic and conventional products are compared, a purely normative (standards-based) comparison leads to wrong conclusions. Positive aspects derived from empirical or modelled data and consequential LCAs should be used wherever feasible.

8.3.2.9 Bias of complexity and data availability

As discussed in relation to sustainable development, impact categories that are difficult to assess tend to be neglected or ignored. This results in the systematic trend that important environmental impacts are not considered. For instance, most assessments of the global warming potential do not include carbon sequestration in the soil, although carbon stores in agricultural soils have been identified as an important factor for climate change. We suggest including these factors as much as possible and applying in-depth sensitivity or uncertainty analysis.

Although the general use of the life cycle assessment method is seen as a significant improvement for assessing environmental impacts of agriculture, if the above implications are not taken into account, we risk biased results of current environmental impact assessments. We, therefore, recommend further research in order to consider the above mentioned problems, which need further methodological developments for a sound comparison of environmental impacts of organic and conventional products.

8.4 Conclusions

Figure 8.6 provides a qualitative overview of environmental impacts of organic farming relative to conventional farming on the basis of the review in Sect. 8.2 and taking into account the methodological discussion in Sect. 8.3. The dots represent the most frequently found result in literature. Because of regional differences, farm and management-specific impacts, and gaps in scientific measurement methodologies, there is a range of uncertainty (blue and grey color). However, several impacts can be determined relatively precisely, because their systematic influence dominates regional or farm-specific differences. The following section will summaries the conclusions regarding the different impact categories.

Impacts of organic farming on **biodiversity** range from much better to equal compared with nonorganic agriculture. **Genetic diversity** can be influenced both

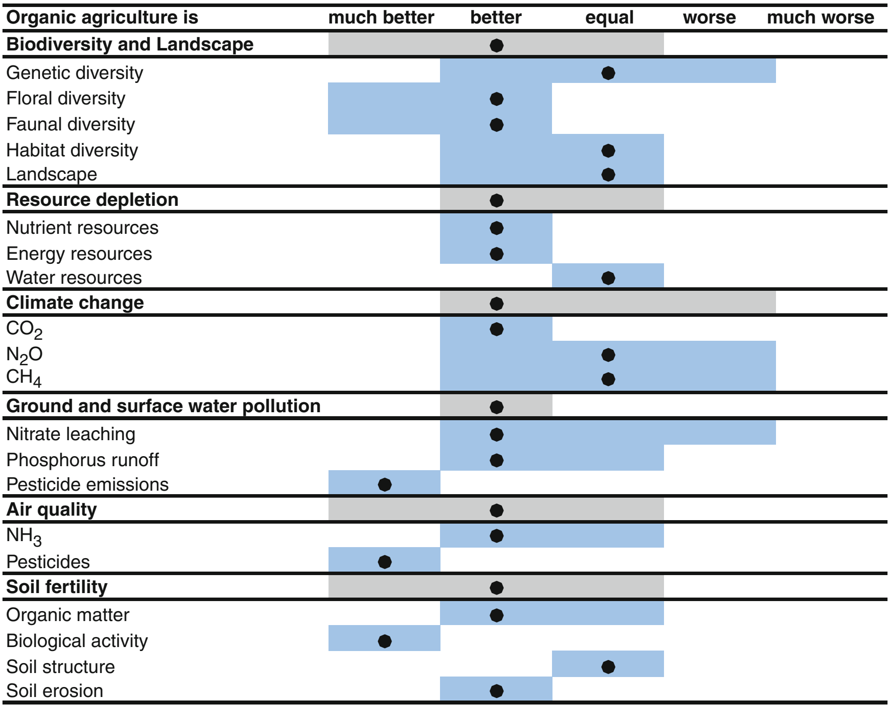

Fig. 8.6 Classification of environmental impacts and relative performance of organic farming compared to conventional farming (Source: Stolze et al. 2000, updated)

positively and negatively in organic systems. Because of lack of scientific evidence, we conclude that both systems perform equally well. According to most studies, organic agriculture clearly performs better for faunal **and floral species diversity** (Bengtsson et al. 2005). Concerning **landscape** and **habitat diversity**, organic farming may perform better because of more diverse crop rotations (Norton et al. 2009) and higher implementation rates of structural elements such as hedges and fruit trees (Schader et al. 2009). However, landscape effects are very farm and site-specific. Therefore, no general trend can be determined (Steiner 2006).

Regarding **resource depletion**, organic farming performs better regarding **nutrients** and **energy**, which confirms the evaluation done by Stolze et al. (2000). Compared with conventional farming, water consumption is not substantially affected by organic farming systems.

Stolze et al. (2000) found that organic agriculture bears a lower CO_2 and NH_3 gas emission potential than conventional farming systems. Furthermore, several long-term trials from the United States, Germany, and Switzerland (Mäder et al. 2002) show that organic farming systems are able to sequester on average 590 kg/ha per year more carbon from the atmosphere than the best performing conventional counterparts. On the basis of the suggested methodological implications, organic farming is likely to perform generally better in terms of CO_2 emissions. Regarding both CH_4 and N_2O emissions, there is not enough scientific evidence to make a final judgement. Furthermore, there are indications of better performance regarding CO_2 sequestration. Thus recent studies suggest a change in the appraisal that Stolze et al. made in 2000 from "equal" to "better."

Eutrophication of **ground and surface water** is very much dependent on what exactly is the subject of comparison. The impacts of nitrate leaching from organic farming can range from better to worse, compared to conventional agriculture. However, most of the studies analyzed found that organic farming performs better. Regarding pesticide emissions into ground and surface water, organic agriculture performs much better due to the ban on artificial pesticides.

Ammonia emissions into the **air** are lower in organic systems, mainly owing to the lower amounts of nitrogen in the system. However, depending on the assumptions, some studies show an equal performance of both systems. Because of the ban of artificial pesticides, air pollution is also lower.

Organic farming performs much better in terms of **soil** biological activity than nonorganic farming. Soil erosion and organic matter content are also affected positively by organic practices, although soil structure remains unaffected.

Both in organic and conventional farming there is potential for improving the environmental performance. Neither of the systems currently satisfies the principles of sustainability. However, organic agriculture on average performs better regarding most of the indicators than conventional systems. Furthermore, if social and economic indicators are taken into account, organic farming seems to render further benefits for society.

Methodologically, comparisons between organic farming need to be adequate to the aims of the study. As has been shown, there are several different levels of comparisons regarding scope, functional unit, stage of processing, life cycle perspective, normativeness, time perspective, scope of impacts, view on sustainability, and units of impacts. In our perception nine severe problems are commonly found in agricultural LCAs and studies using similar approaches. Besides heterogeneity of production systems, these problems are responsible for the highly fluctuating results found in the literature. We suggest reflecting methodological choices more carefully, to be aware of both the uncertainty of the results and the potentially misleading political consequences in the communication of LCA results on agricultural enterprises to consumers and policy makers. Future studies should furthermore integrate the analysis of environmental problems in the socio-economic context in order to come to a comprehensive view of sustainability of farming systems.

References

Aeschenbacher, S., and R. Badertscher. 2008. *Umweltziele Landwirtschaft – hergeleitet aus bestehenden rechtlichen Grundlagen*. Bern: Umweltwissen. Federal Office for the Environment (BAFU), Federal Office for Agriculture (BLW).

Auerswald, K., M. Keinz, and P. Fiener. 2003. Soil erosion potential of organic versus conventional farming evaluated by USLE modelling of cropping statistics for agricultural districts in Bavaria. *Soil Use and Management* 19: 305–311.

Badgley, C., J. Moghtader, E. Quinterio, E. Zakem, M.J. Chappell, K. Avilés-Vázquez, A. Samulon, and I. Perfecto. 2006. Organic agriculture and the global food supply. *Renewable Agriculture and Food Systems* 22(2): 86–108.

Bengtsson, J., J. Ahnström, and A.C. Weibull. 2005. The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology* 42: 261–269.

Boutin, C., A. Baril, and P.A. Martin. 2008. Plant diversity in crop fields and woody hedgerows of organic and conventional farms in contrasting landscapes. *Agriculture, Ecosystems and Environment* 123(1–3): 185–193.

Burton, R.J.F., C. Kuczera, and G. Schwarz, 2008. 'Exploring Farmers' Cultural Resistance to Voluntary Agri-environmental Schemes'. *Sociologia Ruralis* 48: 1, pp. 16–37.

Butler, G., J.H. Nielsen, T. Slots, C. Seal, M.D. Eyre, R. Sanderson, and C. Leifert. 2008. Fatty acid and fat-soluble antioxidant concentrations in milk from high- and low-input conventional and organic systems: Seasonal variation. *Journal of the Science of Food and Agriculture* 88(8): 1431–1441.

Cederberg, C., and M. Stadig. 2003. System expansion and allocation in life cycle assessment of milk and beef production. *The International Journal of Life Cycle Assessment* 6: 350–356.

Chirinda, N., M.S. Carter, K.R. Albert, P. Ambus, J.E. Olesen, J.R. Porter, and S.O. Petersen. 2010. 'Emissions of nitrous oxide from arable organic and conventional cropping systems on two soil types'. *Agriculture, Ecosystems and Environment* 136, pp. 199–208.

Christie, M., N. Hanley, J. Warren, K. Murphy, R. Wright, and T. Hyde. 2006. Valuing the diversity of biodiversity. *Ecological Economics* 58: 304–317.

Condron, L.M., K.C. Cameron, H.J. Di, T.J. Clough, E.A. Forbes, R.G. McLaren, and R.G. Silva. 2000. A comparison of soil and environmental quality under organic and conventional farming systems in New Zealand. *New Zealand Journal of Agricultural Research* 43: 443–466.

Delbecq, A.L., A.H. Van de Ven, and D.H. Gustafson. 1975. *Group techniques for program planning: A guide to nominal group and Delphi processes*, Longman higher education. Glenview: Scott, Foresman.

Edwards, C.A., R. Lal, P. Madden, R.H. Miller, and G. House. 1990. Research on integrated arable farming and organic mixed farming in the Netherlands. In *Sustainable agricultural systems*, ed.

C.A. Edwards, United States. Agency for International Development, et al. Ankeny: Soil and Water Conservation Society.

Eltun, R. 1995. Comparisons of nitrogen leaching in ecological and conventional cropping systems. *Biological Agriculture and Horticulture* 11: 103–114.

Flessa, H., R. Ruser, P. Dörsch, T. Kamp, M.A. Jimenez, J.C. Munch, and F. Beese, 2002. 'Integrated evaluation of greenhouse gas emissions from two farming systems in southern Germany. Special consideration of N2O emissions'. *Agriculture Ecosystems and Environment* 91, pp. 175–199.

Frischknecht, R., H.-J. Althaus, G. Doka, R. Dones, T. Heck, S. Hellweg, R. Hischier, N. Jungbluth, T. Nemecek, G. Reblitzer, and M. Spielmann. 2007. *Overview and methodology*, Final report ecoinvent v2.0, vol. 1. Duebendorf: Swiss Centre for Life Cycle Inventories.

Fuller, R.J., L.R. Norton, R.E. Feber, P.J. Johnson, D.E. Chamberlain, A.C. Joys, F. Mathews, R.C. Stuart, M.C. Townsend, W.J. Manley, M.S. Wolfe, D.W. Macdonald, and L.G. Firbank. 2005. Benefits of organic farming to biodiversity vary among taxa. *Biology Letters* 1: 431–434.

Gattinger, A., A. Müller, M. Häni, B. Oehen, and U. Niggli, 2010. *Klimaleistungen des Biolandbaus - Fakten und Hintergründe* Frick.

Gibson, R.H., S. Pearce, R.J. Morris, W.O.C. Symondson, and J. Memmott. 2007. Plant diversity and land use under organic and conventional agriculture: A whole-farm approach. *Journal of Applied Ecology* 44(4): 792–803.

Goulding, K.W.T. 2000. Nitrate leaching from arable and horticultural land. *Soil Use and Management* 16: 145–151.

Grönroos, J., J. Seppälä, P. Voutilainen, P. Seuri, and K. Koikkalainen. 2006. Energy use in conventional and organic milk and rye bread production in Finland. *Agriculture, Ecosystems & Environment* 117(2–3): 109–118.

Gruber, W. 1992. *Die schweizerische Agrarpolitik vor einer ökologischen Neuorientierung?* Freiburg: Universitätsverlag Freiburg.

Haas, G., F. Wetterich, and U. Köpke. 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, Ecosystems and Environment* 83: 43–53.

Halberg, N. 2008. Energy use and green house gas emission in organic agriculture. *Paper presented at the international conference organic agriculture and climate change*, Enita of Clermont, France 17–18 April 2008.

Hansen, S., M.A. Bleken and B.K. Sitaula. 2008. Effect of soil compaction and fertilization practise on N_2O emission and CH_4 oxidation (Presentation). International Symposium "Organic Farming and Climate Change" 17 and 18 April 2008 at Clermont-Ferrand ENITA/France (http://www.abiodoc.com/fileadmin/uploads/Colloque/Diaporama/TH05/18_Hansen.pdf).

Heijungs, R., J.B. Guinée, G. Huppes, R.M. Lankreijer, H.A.U. de Haes, and A.W. Sleeswijk. 1992. *Environmental life cycle assessment of products*. Leiden: Centre of Environmental Science.

Herzog, F., and W. Richner. 2005. *Evaluation der ökomassnahmen—bereich stickstoff und phosphor*, Schriftenreihe der FAL, vol. 57. Reckenholz: Agroscope FAL Reckenholz.

Hoeppner, J.W., M.H. Entz, B.G. McConkey, R.P. Zentner, and C.N. Nagy. 2005. Energy use and efficiency in two Canadian organic and conventional crop production systems. *Renewable Agriculture and Food Systems* 21(1): 60–67.

Hole, D.G., A.J. Perkins, J.D. Wilson, I.H. Alexander, P.V. Grice, and A.D. Evans. 2005. Does organic farming benefit biodiversity? *Biological Conservation* 122: 113–130.

Hospido, A., J. Davis, J. Berlin, and U. Sonesson. 2010. A review of methodological issues affecting LCA of novel food products. *International Journal of Life Cycle Assessment* 15: 44–52.

Huber, B., O. Schmid, and G. Napo-Bitanem. 2010. Standards and regulations. *The world of organic agriculture—Statistics and emerging trends*, IFOAM, Bonn; FiBL, Frick; ITC, Geneva. 72–79.

IFOAM. 2009. International federation of organic agriculture movements (ifoam). http://www.ifoam.org/index.html. Accessed 5 Dec 2009.

IPCC. 2007. *Climate change 2007—Synthesis report*. Cambridge: Intergovernmental Panel on Climate Change (IPCC).

Jeanneret, P., D. Baumgartner, R. Freiermuth, and G. Gaillard. 2006. *Méthode d'évaluation de l'impact des activités sur la biodiversité*. Zürich-Reckenholz: Agroscope Reckenholz Tänikon (ART).

Jeffreys, I. 2002. A multi-objective decision-support system (modss) with stakeholders and experts in the hodgson creek catchment. In *A whole-farm and regional agroforestry decision-support system. A report for the rirdc/land & water Australia/FWPRDC joint venture agroforestry program*, eds. Harrison S., and J. Herbohn. Rural Industries Research and Development Corporation (RIRDC).

Kirchmann, H., and L. Bergström. 2001. Do organic farming practices reduce nitrate leaching? *Communications in Soil Science and Plant Analysis* 32(7): 997–1028.

Knop, E., D. Kleijn, F. Herzog, and B. Schmid. 2006. Effectiveness of the swiss agri-environment scheme in promoting biodiversity. *Journal of Applied Ecology* 43: 120–127.

Köpke, U. 2003. Conservation agriculture with and without use of agrochemicals. *Paper presented at the II. World congress on conservation agriculture "Producing in Harmony with Nature,"* Foz do Iguassu, Brazil, August 11–15, 2003.

Köpke, U., E.R. Keller, and G. Kahnt. 1997. Ökologischer landbau. In *Handbuch des pflanzenbaus 1. Grundlagen der landwirtschaftlichen pflanzenproduktion*, vol. Kapitel 9, ed. K.-U. Heyland, H. Hanus, E.R. Keller, et al. Stuttgart: Verlag Eugen Ulmer.

Krystallis, A., and G. Chryssohoidis. 2005. Consumers' willingness to pay for organic food: Factors that affect it and variation per organic product type. *British Food Journal* 107(5): 320–343.

Küstermann, B., M. Kainz and K.-J. Hülsbergen. 2008. Modeling carbon cycles and estimation of greenhouse gas emissions from organic and conventional farming systems. Renewable Agriculture and Food Systems 23(1): 38–52.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304(5677): 1623–1627.

Lampkin, N.H. 1990. *Organic farming*. Ipswich: Farming Press Books.

Lampkin, N.H. 2007. Organic farming's contribution to climate change and agricultural sustainability. *Paper presented at the Welsh organic producers conference*, 18th Oct 2007.

Lynch, D. 2008. Greenhouse gas emissions from cropping in Atlantic Canada (Presentation). International Symposium "Organic Farming and Climate Change", 17 and 18 April 2008 at Clermont-Ferrand ENITA/France (http://www.abiodoc.com/fileadmin/uploads/Colloque/Diaporama/TH05/21_Lynch.pdf).

Mäder, P., A. Fließbach, D. Dubios, L. Gunst, P. Fried, and U. Niggli. 2002. Soil fertility and biodiversity in organic farming. *Science* 296: 1694–1697.

Meadows, D.H., D.I. Meadows, J. Randers, and W.W. Beherns III. 1972. *The limits to growth—A report to the club of Rome*. New York: Universe Books.

Morris, C., A. Hopkins, and M. Winter. 2001. *Comparison of the social, economic and environmental effects of organic, ICM and conventional farming—Final report to the countryside agency*. Cheltenham, IGER: CCRU.

Nemecek, T., O. Huguenin-Elie, D. Dubois, and G. Gaillard. 2005. *Ökobilanzierung von anbausystemen im schweizerischen acker- und futterbau*, FAL Schriftenreihe, vol. 58. Reckenholz: Eidgenössische Forschungsanstalt für Agrarökologie und Landbau (FAL).

Niggli, U. 2007. The evolution of organic practice. In *Organic farming: An international history*, ed. W. Lockeretz. Oxfordshire, Cambridge: CAB International.

Niggli, U., A. Fließbach, M. Stolze, J. Sanders, C. Schader, G. Wyss, O. Balmer, L. Pfiffner, and E. Wyss. 2008. *Gesellschaftliche leistungen der biologischen landwirtschaft*. Frick: Forschungsinstitut für Biologischen Landbau (FiBL).

Niggli, U., A. Fliessbach, P. Hepperly, and N. Scialabba. 2009. *Low greenhouse gas agriculture: Mitigation and adaptation potential of sustainable farming systems*. Rome: Food and Agriculture Organization of the United Nations (FAO).

Norton, L., P. Johnson, A. Joys, R. Stuart, D. Chamberlain, R. Feber, L. Firbank, W. Manley, M. Wolfe, and B. Hart. 2009. Consequences of organic and non-organic farming practices for field, farm and landscape complexity. *Agriculture, Ecosystems and Environment* 129(1–3): 221–227.

OECD. 2001. *Multifunctionality—Towards an analytical framework*. Paris: Organisation for Economic Co-operation and Development (OECD).

OECD. 2004. *Workshop on evaluating agri-environmental policies*. Paris: Organisation for Economic Cooperation and Development (OECD).

Olesen, J.E., K. Schelde, A. Weiske, M.R. Weisbjerg, W.A.H. Asman, and J. Djurhuus. 2006. Modelling greenhouse gas emissions from european conventional and organic dairy farms. *Agriculture, Ecosystems & Environment* 112(2–3): 207–220.

Osterburg, B., and T. Runge. 2007. *Maßnahmen zur reduzierung von stickstoffeinträgen in gewässer—eine wasserschutzorientierte landwirtschaft zur umsetzung der wasserrahmenrichtline*, vol. 307. Braunschweig: Bundesforschungsanstalt für Landwirtschaft (FAL), Landbauforschung Völkenrode—FAL Agricultural Research.

Pacini, C., A. Wossink, G. Giesen, C. Vazzana, and R. Huirne. 2003. Evaluation of sustainability of organic, integrated and conventional farming systems: A farm and field-scale analysis. *Agriculture, Ecosystems & Environment* 95(1): 273–288.

Petersen, S.O., K. Regina, A. Pöllinger, E. Rigler, L. Valli, S. Yamulki, M. Esala, and C. Fabbri, E. Syväsalo and F.P. Vinther. 2006. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. Agriculture, Ecosystems and Environment 112: 200–206.

Pfiffner, L., and H. Luka. 2007. Earthworm populations in two low-input cereal farming systems. *Applied Soil Ecology* 37: 184–191.

Pimentel, D., and M.H. Pimentel. 2005. Sustainability of meat-based and plant-based diets and the environment. *The American Journal of Clinical Nutrition* 78: 660–663.

Pimentel, D., C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Christ, L. Shpritz, L. Fitton, R. Saffouri, and R. Blair. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* 267: 1117–1123.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. *Bioscience* 55(7): 573–582.

Reap, J., F. Roman, S. Diuncan, and B. Bras. 2008a. A survey of unresolved problems in life cycle assessment—Part 1: Goal and scope and inventory analysis. *The International Journal of Life Cycle Assessment* 13: 290–300.

Reap, J., F. Roman, S. Diuncan, and B. Bras. 2008b. A survey of unresolved problems in life cycle assessment—Part 2: Impact assessment and interpretation. *The International Journal of Life Cycle Assessment* 13: 374–388.

Sanders, J., N.H. Lampkin, P. Midmore, and M. Stolze. 2005. A policy impact model for organic farming in Switzerland. In *ISOFAR: Proceedings of the conference "Researching Sustainable Systems", Adelaide 2005*, ed. U. Köpke, U. Niggli, D. Neuhoff, P. Cornish, W. Lockeretz, and H. Willer. Bonn: International Society of Organic Agriculture Research (ISOFAR).

Schader, C. 2009. *Cost-effectiveness of organic farming for achieving environmental policy targets in Switzerland*. Frick: Aberystwyth University, Wales. Research Institute of Organic Farming (FiBL).

Schader, C., and M. Stolze. 2010. *Nachhaltigkeitsbewertung der knospe-landwirtschaft—ergebnisse eines expertenworkshops*. Frick: Research Institute of Organic Agriculture (FiBL).

Schader, C., T. Nemecek, G. Gaillard, J. Sanders, and M. Stolze. 2008a. Using LCA data for agri-environmental policy analysis at sector level. *Paper presented at the 6th international conference on life cycle assessment in the agri-food sector 2008—Towards a sustainable management of the food chain*, Zurich, Switzerland, 12–14 Nov 2008.

Schader, C., L. Pfiffner, C. Schlatter, and M. Stolze. 2008b. Umsetzung von ökomassnahmen auf bio- und öln-betrieben. *Agrarforschung* 15(10): 506–511.

Schader, C., J. Sanders, T. Nemecek, N.H. Lampkin, and M. Stolze. 2008c. A modelling approach for evaluating agri-environmental policies at sector level. *Yearbook of Socioeconomics in Agriculture* 1: 93–132.

Schader, C., L. Pfiffner, C. Schlatter, and M. Stolze. 2009. Umsetzung von Agrarumweltmassnahmen auf bio- und konventionellen Betrieben der Schweiz. In: Mayer, J., Alföldi, T., Leiber, F., et al. eds. Werte—Wege—Wirkungen: Biolandbau im Spannungsfeld zwischen Ernährungssicherung, Markt und Klimawandel. Beiträge zur 10. Wissenschaftstagung Ökologischer Landbau, 11–13

Feb 2009, Zürich, 2009. Forschungsanstalt Agroscope Reckenholz-Tänikon (ART), Forschungsinstitut für biologischen Landbau (FiBL), ETH Zürich, Stiftung Ökologie & Landbau, 446–449.

Schmidt, J. 2008. System delimitation in agricultural consequential LCA. *The International Journal of Life Cycle Assessment* 13: 350–364.

Sehy, U. 2004. *N2O-Freisetzungen aus Ackerböden*: PhD Thesis. Technical University Munich.

Shepherd, M.A., R. Harrison, and J. Webb. 2002. Managing soil organic matter implications for soil structure on organic farms. *Soil Use and Management* 18: 284–292.

Shepherd, M., B. Pearce, B. Cormack, L. Philipps, S. Cuttle, A. Bhogal, P. Costigan, and R. Unwin. 2003. *An assessment of the environmental impacts of organic farming*. London: DEFRA, ADAS, Elm Farm, IGER.

Siegrist, S., D. Schaub, L. Pfiffner, and P. Mäder. 1998. Does organic agriculture reduce soil erodibility? The results of a long-term field study on loess in Switzerland. *Agriculture, Ecosystems & Environment* 69: 253–265.

Smith, P., D. Martino, Z. Cai, H. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. *Agriculture*, Climate Change 2007: Mitigation. Cambridge: Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Steiner R.S. 2006. Landnutzungen prägen die Landschaft—konventionelle, bio-organische und bio-dynamische Anbaumethoden im Vergleich in ihrer Wirkung auf die Agrarlandschaft im Kanton Zürich.

Stern, N. 2007. *The economics of climate change—The Stern review*. London: Cambridge University Press.

Stockdale, E.A., N.H. Lampkin, M. Hovi, R. Keatinge, E.K.M. Lennartsson, D.W. Macdonald, S. Padel, F.H. Tattersall, M.S. Wolfe, and C.A. Watson. 2001. Agronomic and environmental implications for organic farming systems. *Advances in Agronomy* 70: 261–327.

Stolze, M., A. Piorr, A.M. Häring, and S. Dabbert. 2000. *Environmental impacts of organic farming in Europe*, Organic farming in Europe: Economics and policy, vol. 6. Stuttgart-Hohenheim: Universität Stuttgart-Hohenheim.

Stopes, C., E.I. Lord, L. Philipps, and L. Woodward. 2002. Nitrate leaching from organic farms and conventional farms following best practice. *Soil Use and Management* 18: 301–308.

Stotten, F. 2008. *Akzeptanzanalyse von naturschutzmaßnahmen im rahmen der schweizerischen agrarpolitik*. Aachen: RWTH Aachen.

TEEB (2010). *The Economics of Ecosystem and Biodiversity: Ecological and economic foundations.* In Kumar, P. (ed.), London and Washington: Earthscan.

Thomassen, M.A., R. Dalgaard, R. Heijungs, and I. de Boer. 2008a. Attributional and consequential lca of milk production. *The International Journal of Life Cycle Assessment,* 13(339–349).

Thomassen, M.A., K.J. van Calker, M.C.J. Smits, G.L. Iepema, and I.J.M. de Boer. 2008b. Life cycle assessment of conventional and organic milk production in the Netherlands. *Agricultural Systems* 96: 85–107.

UNECE. 1999. *Gothenburg protocol to abate acidification, eutrophication and ground-level ozone*. Gothenburg: United Nations Economic Commission for Europe (UNECE).

Vogt, G. 2007. The origins of organic farming. In *Organic farming: An international history*, ed. W. Lockeretz. Oxfordshire, Cambridge: CAB International.

Willer, H. and L. Kilcher. 2009. *The World of Organic Agriculture, Statistics and Emerging Trends 2009*: IFOAM, Bonn; FiBL, Frick; ITC, Geneva.

Woese, K., D. Lange, C. Boess, and K.W. Bögl. 1997. A comparison of organically and conventionally grown foods—Results of a review of the relevant literature. *Journal of the Science of Food and Agriculture* 74: 281–293.

Younie, D. and C. A. Watson. 1992. Soil Nitrate–N Levels in organically and intensively managed grassland systems. Aspects of Applied Biology, 30: 235–238.

Ziesemer, J. 2007. *Energy use in organic food systems*. Rome: Natural Resources Management and Environmental Department, Food and Agriculture Organisation of the United Nations (FAO).

Zittel, W., and J. Schindler. 2007. *Crude oil: The supply outlook*. Berlin: Energy Watch Group.