

Environmental Impact of Organic Agriculture

K. Lorenz¹, R. Lal

Carbon Management and Sequestration Center, School of Environment and Natural Resources, College of Food, Agricultural, and Environmental Sciences, The Ohio State University, Columbus, OH, USA

¹Corresponding author. E-mail address: lorenz.59@osu.edu

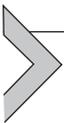
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Abstract

Organic agriculture (OA) is practiced on 1% of the global agricultural land area and its importance continues to grow. Specifically, OA is perceived by many as having less

negative effects on the environment than conventional agriculture because applications of soluble mineral fertilizers, and synthetic herbicides and pesticides are prohibited. However, scientific evidence for better environmental impact is scanty. Specifically, yields under OA are about 19% lower and the attendant lower soil carbon (C) inputs together with tillage for weed control contributes to lower profile soil organic carbon (SOC) stocks under OA. Less well known are the effects on soil inorganic carbon (SIC) stocks. Otherwise, soils managed by OA may emit less carbon dioxide (CO₂), nitrous oxide (N₂O), and methane (CH₄). Specifically, by the adoption of OA practices 1.65 Mg CO₂ ha⁻¹ y⁻¹ may be sequestered in the top 20-cm layer. Further, N₂O emissions from soils managed by OA may be 492 kg CO₂ eq. ha⁻¹ y⁻¹ lower than those from conventionally managed soils. Under OA management, a higher CH₄ uptake of 3.2 kg CO₂ eq. ha⁻¹ y⁻¹ may be observed for arable soils. The soil, air, and water quality may also be enhanced by OA whereas effects on biodiversity are mixed. Thus, there is an urgent need to strengthen the database on environmental impacts of OA by establishing and studying long-term field experiments in all major biomes and principal soils. Consumer demand for organic products will continue to grow driven by food safety concerns and increasing affluence. Due to lower yields, however, natural ecosystems may be increasingly converted to agroecosystems to meet the demand with less well-known consequences for the environment. Nonetheless, scientific interest in OA is less than a century old, and there is significant potential to lessen its environmental impacts while methods derived from OA can contribute to sustainable intensification of agricultural systems.



1. INTRODUCTION

Agriculture has a major global impact as ~40% of the global ice-free land area is already under agricultural production (Ramankutty et al., 2008). Specifically, ~12% of the global ice-free land area is covered by croplands and an additional ~28% by grasslands including rangelands, shrublands, pastureland, and cropland sown with pasture and fodder crops (Conant, 2012; Ramankutty et al., 2008). Agriculture supports the livelihoods and subsistence of the largest number of people worldwide, and is vital to rural development and poverty alleviation, as well as to food and nonfood production (WBCSD, 2008). The main challenges for the agricultural sector are to: (1) simultaneously secure enough high-quality agricultural production to meet increasing demand; (2) conserve biodiversity and manage natural resources; and (3) improve human health and well-being, especially for the rural poor in developing countries (WBCSD, 2008). However, the current agricultural land use practices have already substantial environmental impacts such as biodiversity loss, accelerated soil erosion and degradation,

eutrophication including algal blooms and oceanic dead zones, pesticide effects on humans and wildlife, greenhouse gas (GHG) emissions, and regime shifts in hydrological cycling (Ponisio et al., 2015). If current trends in population growth, food and energy consumption, and food waste continue, the problems of hunger, food insecurity, and environmental degradation will be drastically exacerbated.

Conventional approaches to intensify agriculture and, in particular, the unbridled use of irrigation and fertilizers are among the major causes of environmental degradation (Foley et al., 2011). Thus, sustainable intensification (SI) of agriculture has been proposed to reduce the negative biophysical impacts of modern agricultural practices (Garnett et al., 2013). The goal of SI is to optimize crop production per unit area while accounting for social, political, and environmental impacts (Bennett et al., 2014). Therefore, the focus is on increased production efficiency at lower environmental and resource costs. Examples for SI practices include using improved irrigation techniques that give more crop-per-drop, increasing yield per unit input, adopting climate-smart agriculture that produces less GHG per unit product, reducing use of energy by using conservation agriculture, and recycling nutrients. Many of these practices aim to achieve and maintain the highest possible productivity at a given location for the lowest economic and environmental cost (Bennett et al., 2014). However, more sustainable and more resilient agricultural practices may produce lower yields compared to current practices, and more land would be needed to produce the same amount of food. To meet future global demands, food waste and meat consumption must also be reduced, and the distribution of food improved (Foley et al., 2011). Thus, more than mere changes in agricultural production systems are required and equally radical agendas must be pursued to reduce resource-intensive consumption and waste, and to improve governance, efficiency, and resilience (Garnett et al., 2013).

Instead of chemically intensive and biologically simplified systems, cost-effective cultivation techniques are needed that encourage ecological interactions to generate soil fertility, nutrient cycling and retention, water storage, pest/disease control, pollination, and other essential agricultural inputs/ecosystem services (Kremen et al., 2012). In particular, some studies of organic agriculture (OA) indicate better performance than that of conventional systems with regard to species richness and abundance, soil fertility, nitrogen uptake by crops, water infiltration rate and holding

capacity, and energy use and efficiency (Ponisio et al., 2015). Critical to the success of OA are cover crops and green manures because these offer multiple essential functions including fixing nitrogen (N), adding organic matter (OM), and providing habitat for beneficial organisms (Abbott and Manning, 2015). While many explanations and definitions for OA exist, common technologies to maintain soil fertility and produce high-quality products in OA are: (1) applying appropriate rotation programs, (2) adding composts, (3) using physical, mechanical, biological mechanism to control diseases, pests, and weeds, and (4) adopting organic methods in the feed and livestock production (Shi-ming and Sauerborn, 2006). Thus, approaches for improving the production, food security, and environmental performance of agriculture should also include the adaptation of lessons from OA (Foley et al., 2011). To produce sufficient food supply for a growing world population while minimizing the negative environmental impact, further improvements of conventional agriculture based on innovations, enhanced efficiency, and improved agronomic practices seem to be the only way (Kirchmann et al., 2008a).

In the context of modern agriculture, the first distinct form of OA was introduced in 1924 by Rudolf Steiner's course on Social Scientific Basis of Agricultural Development introducing the concept of the farm as an organism (Table 1; Stockdale et al., 2001). Steiner's lectures formed the basis of biological dynamic (biodynamic) agriculture which was developed toward the end of the 1920s in Germany, Switzerland, England, Denmark, and the Netherlands. Since then, research and practice of biodynamic, and those of organic, biological organic, and modern OA expanded worldwide (Kirchmann et al., 2008b). The underpinning principles of these practices regarding exclusion of synthetic compounds (fertilizers and pesticides) is still the main driver for choosing crops and pest control methods in OA. However, although a fast growing sector, only about 1% of the worldwide agricultural land area is managed by OA practices during the decade of 2010s (Willer and Lernoud, 2015).

Consumers increasingly demand accessible, environment-friendly, nutritional, and safe food (Lairon and Huber, 2014). However, nutritional and toxicological value of food produced under OA methods of production, as well as their potential effects on animal and human health are uncertain. The consumption of organic food may reduce the exposure to pesticide residues (Barański et al., 2014). However, a direct cause-effect relationship between organic food consumption and consumer's health has not been established (Smith-Spangler et al., 2012). Furthermore, OA has other environmental

Table 1 Forms of organic agriculture, their philosophies/characteristics, and management implications [all forms exclude synthetic fertilizers and pesticides; [Kirchmann et al. \(2008b\)](#)]

| Form | Philosophies/principles | Management implications |
|------------------------------------|---|--|
| Biological dynamic (biodynamic) | Only natural products contain curing and saving forces Organic food provides spiritual forces to mankind Farms are closed entities and self-sustaining units | Management of forces related to spiritual matters that act in soil, crops, and animals Application of mixtures of minerals, wild plants, and animal organs to soil, crop, and animal manures Crop sowing or planting according to astrological principles |
| Organic | Close relationship between soil fertility and human health Food quality is important for human health | Essential aim is to maintain or increase soil organic matter contents as guarantee of soil health Only composted organic materials should be applied to maintain soil fertility as synthetic fertilizers speed up the rate at which soil organic matter is exhausted |
| Biological organic | Healthy soils are the basis for health on earth Soil humus is the most significant of all nature's reserves Applying nature's principles through analogical, biological thinking Recognizing biological wholeness with a holistic view on food production and nature Humus formation is a sign of fertility and not humus as such Humification is the greatest biological regulation known to nature | Normal humus formation is only achieved when natural soil layering is not disturbed Soil tillage should be kept at a minimum to avoid disorder in soil layering Organic manures and composts must only be used as surface cover as they are not suitable for the root zone |

(Continued)

Table 1 Forms of organic agriculture, their philosophies/characteristics, and management implications [all forms exclude synthetic fertilizers and pesticides; Kirchmann et al. (2008b)]—cont'd.

| Form | Philosophies/principles | Management implications |
|----------------|--|--|
| Modern organic | <p>Application of soluble salts to soil does not fulfil the demands of crops</p> <p>Nutrient supply is not synchronized with the growth of crops</p> <p>Losses of nutrients are inevitable and high from artificial fertilizers compared with organic manures because the organic but not the artificial fertilizer is adapted to the turnover in soil</p> | <p>Exclusion of synthetic compounds</p> <p>Use of natural means and methods only</p> <p>Production is based on ecological processes and recycling, and should fit the cycles and ecological balance in nature</p> <p>Management in a precautionary and responsible manner to protect the health and well-being of current and future generations and the environment</p> |
| | <p>Health chain from soils that produce healthy crops, fostering health of animals and humans</p> <p>Principle of fairness—respect, justice, eradication of poverty, animal welfare, equitable systems for distribution and trade, as well as social costs</p> | |
| | <p>Science is necessary to ensure that organic agriculture is healthy, safe, and ecologically sound</p> | |
| | <p>Scientific knowledge alone is not sufficient</p> | |
| | <p>Practical experience, accumulated wisdom, and traditional and indigenous knowledge offer valid solutions, tested by time</p> | |
| | | |

impacts similar to those of conventional land uses. To reduce those impacts, the International Federation for Organic Agricultural Movements (IFOAM) requires organic farms to avoid all forms of pollution, and to maintain the genetic diversity of the agricultural system and its surroundings, including the protection of plant and wildlife habitats (Stockdale et al., 2001). On the positive side, OA may enhance soil fertility, nutrient cycling and retention, water storage, pest/disease control, pollination, and other essential positive agricultural inputs/ecosystem services. Similar to conventional agricultural practices, however, negative environmental impacts by OA may arise from: (1) utilization of animal manures, (2) use of natural fertilizers and pesticides, (3) management of postharvest residues, (4) irrigation, and (5) tillage operations (Udeigwe et al., 2015). Among the major environmental impacts may be contamination of soil, water, and air by nutrients, organic carbon (C), heavy metals, and pathogens, as well as air contamination by particulate matters, noxious gases, and pathogens. Nevertheless, certain OA practices (eg, the application of animal manures, crop residue handling, and irrigation water use) may be sources as well as facilitators of the transport of the aforementioned pollutants within the environment (Udeigwe et al., 2015).

Several health-related issues in humans have been attributed to a number of agricultural pollutants, some of them also attributed to OA. For example, linkages between respiratory diseases and particulate matters (PM_{2.5} and PM₁₀) (Arbex et al., 2007), have been widely documented. Likewise, a number of human health issues relating to trace element (eg, copper) ingestion have been reported (Uriu-Adams and Keen, 2005). Pathogens present in animal manure can cause a number of health problems in humans (Mathis et al., 2005), such as environmental contamination by nutrients (eg, phosphorus) from agricultural sources with adverse effects on human health (Fawell and Nieuwenhuijsen, 2003; Kalantar-Zadeh et al., 2010). However, there is no consistent evidence regarding the health status of farm workers in relation to OA. Contrarily, there is some evidence that pesticides applied in conventional agriculture contribute to genetic damages in farm workers (Costa et al., 2014). Therefore, additional research is required on health effects of farming systems on farmers.

The current state of OA is briefly discussed in the following section, specific effects of OA on soil C stocks, soil-derived greenhouse gases (GHGs), and yield are discussed in detail in subsequent sections. The general implications of OA for the environment are presented in the concluding section.



2. CURRENT STATE OF ORGANIC AGRICULTURE

OA is the most rapidly growing, contentious, and innovative farming system which balances several sustainability goals to promote global food and ecosystem security (Crowder and Reganold, 2015). However, the global share of agricultural land under OA is small but the consumer demand for organic food, particularly, in Europe and the United States is growing (Willer and Lernoud, 2015). Specifically, among the 4300 Mha of global agricultural land (Ramankutty et al., 2008), about 43 Mha (1%) was under OA in 2013, including in-conversion areas (Willer and Lernoud, 2015). However, data on land use for OA are only available for 170 countries in the decade of the 2010s. Nevertheless, the global OA area was 6 Mha more in 2013 than that in 2012, mainly as 5 Mha more were reported from Australia where rangeland areas came into organic production. The organic land area has increased in all surveyed regions except in Latin America, because the organic grazing areas decreased in Argentina. In addition to Australia, major increases in areas under OA are reported for China, Italy, Peru, and Ukraine. Aside agricultural land, 35 Mha nonagricultural land (ie, land for wild collection, aquaculture, forests, and grazing areas on nonagricultural land) were organic. In total, the area under organic grassland/grazing was 27 Mha compared with 7.7-Mha arable land under OA. However, details on land use in 2013 were only available for 90% of the OA land (Willer and Lernoud, 2015). Further, more than 11 Mha of agricultural land under OA and more than 1.7 million of organic producers were in developing countries and emerging markets. The data on producers were uncertain as some countries reported only the number of companies, projects, or grower groups. Global sales of organic food reached US \$72 billion in 2013, and revenues have increased almost fivefold since 1999. Europe and North America alone generated over 90% of global sales of organic food. From 2012 to 2013, sales of organic products increased by 6% in Europe and by 11.5% in the United States. Consumer demand for organic food is growing partly due to some concerns about the food safety (Willer and Lernoud, 2015).

Critical to the adoption of OA is the financial competitiveness compared to that of the conventional agriculture (Crowder and Reganold, 2015). For example, when organic premiums are not applied, OA is less profitable than conventional agriculture, that is, benefit/cost ratios (−8 to −7%) and net present values (−27 to −23%) for OA can be lower than those for

conventional agriculture based on a metaanalysis of a global dataset spanning 55 crops grown on 5 continents (Crowder and Reganold, 2015). However, OA can be more profitable (22–35%) and have higher benefit/cost ratios (20–24%) than conventional agriculture when actual premiums are applied. Although actual premiums are 29–32%, breakeven premiums necessary for organic profits to match conventional profits are only 5–7%, even with organic yields being 10–18% lower. Financially, OA may also be favored by lower environmental costs (negative externalities) and enhanced ecosystem services from the adoption of good farming practices (Crowder and Reganold, 2015).

Presently, OA has developed into a highly standardized food production protocol regulated by more than 80 national laws while 16 countries are in the process of drafting legislation (Willer and Lernoud, 2015). In addition, 38 countries have alternative organic certification protocols, that is, locally-focused quality assurance systems (PGS—Participatory Guarantee Systems), and those systems are under development in 17 more countries. About 80% of the organic food is consumed in the US and EU markets, while 75% of the producers produce outside of these two major domestic markets. However, in most European countries, conversion rates of farmers to OA are low although market demand is huge and direct payment schemes support conversion. In export-oriented countries, the growing trade threatens the regionalization and contextualization of OA because the standards of the EU and US markets are the dominant requirements (Willer and Lernoud, 2015).

In summary, the global land area under OA is small but projected to increase, particularly, as demand for organic food continues to grow in Europe and the United States. Even if organic premiums decline, OA can continue to expand, and organic farming systems can contribute a larger share in feeding the world with their multiple sustainability benefits (Crowder and Reganold, 2015). Aside addressing food security, it is increasingly recognized that OA can play a role in addressing land and soil degradation, climate change, poverty alleviation, hunger, health, and biodiversity stewardship (Willer and Lernoud, 2015). Consequential life cycle assessment of agricultural products must be applied (Meier et al., 2015), in order to support policy making and strategic environmental planning toward increased adoption of OA. Scientific evidence of how some of the sustainability challenges, particularly those related to soil processes, are addressed by OA and are discussed in the following sections.



3. EFFECTS OF ORGANIC AGRICULTURE ON SOIL CARBON STOCKS

The C stock in soils of agroecosystems is comprised of the soil organic C (SOC) and the soil inorganic C (SIC) stocks. The SIC stock consists of lithogenic inorganic C (LIC) or primary carbonates derived from the soil parent material, and pedogenic inorganic C (PIC) or secondary carbonates formed through soil processes (Sanderman, 2012). Carbon enters the SOC stock via the inputs of C from photosynthetic fixation of atmospheric carbon dioxide (CO_2) by vegetation, deposition of microbial and plant residues, and organic amendments (animal manure, biosolids). The main C input to soil is the net primary production (NPP) as a major fraction of the CO_2 fixed during plant photosynthesis by gross primary production (GPP), which is respired autotrophically and returned back to the atmosphere. NPP enters the soil by rhizodeposition and decomposition of plant litter, and the major fraction is converted back to CO_2 by soil respiration and some lost as methane (CH_4). Aside microbial decomposition enhanced by soil tillage, C losses from soils of agroecosystems are associated with erosion, fire, harvest, and leaching (Ciais et al., 2010, 2011; Chang et al., 2015). Site-specific factors (eg, climate, physicochemical characteristics, soil and vegetation management) determine the balance between C input and losses. In the following sections, comparisons of the effects of conventional and organic farming systems on SIC and SOC stocks as well as the effects of different OA practices on SOC stocks will be discussed. Research priorities will also be identified to strengthen the knowledge base.

3.1 Soil Inorganic Carbon Stock

In many important agricultural regions, SIC stocks can rival those of SOC (Sanderman, 2012). In arid and semiarid regions, SIC stocks can even be many times greater than those of SOC, making changes in SIC stock as a result of agricultural soil and land-use management practices a potentially major C flux in soils where carbonates are present (Ahmad et al., 2015). In general, SIC refers to mineral carbonates, dominated by calcium carbonate (CaCO_3). However, sodium and magnesium carbonates can also be present in significant quantities in salt-affected soils (Sanderman, 2012). A large fraction of SIC in many soils can be inherited from calcareous parent material (limestone and other marine carbonates), termed primary or lithogenic soil

carbonates. The soil carbonates formed in situ by the precipitation of CaCO_3 are termed secondary or pedogenic carbonates. The latter are formed through the reaction of a dilute carbonic acid formed through dissolution of atmospheric CO_2 in soil with Ca^{2+} and Mg^{2+} brought in from outside the local agroecosystem, for example, with calcareous dust, and by agricultural practices such as irrigation, fertilization including manuring and liming (Lal, 2008). Thus, the prediction of potential responses of soil C to agricultural land-use change and management practices cannot be based entirely on that of SOC as, for example, SIC stocks in agricultural soils may change by as much as $1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ (Sanderman, 2012). However, for the purposes of C sequestration by soil carbonate formation, calcium can be used only once—when it is released from silicates and not when released from pre-existing carbonate (Monger et al., 2015).

Agricultural practices may alter processes which affect SIC fluxes and storage. Among those are enhanced mineral weathering as a result of organic acids present in agricultural soils, calcite precipitation/dissolution, dolomite dissolution, and changes in dissolved CO_2 related to changes in soil pH (Lal, 2008). However, there is no consensus on the impact of land management practices in general and of OA practices in particular, on SIC dynamics. For example, in an irrigated cotton (*Gossypium arboreum* L.) agroecosystem in semiarid New Mexico, USA, SIC stocks to 1-m depth in field plots 3, 6, and 9 years under OA were lower (83.4, 79.8, and 91.6 Mg C ha^{-1} , respectively) than those under conventional agriculture (111.9 Mg C ha^{-1} ; Jacinthe et al., 2011). The fields managed organically for 3 and 9 years were under alfalfa (*Medicago sativa* L.)–cotton rotation, whereas those managed organically for 6 years were primarily under alfalfa, with occasional plantings of corn (*Zeamays* L.), chile (*Capsicum annuum* L.), and lettuce (*Lactuca sativa* L.). All three organic fields were moldboard-ploughed followed by chisel tillage using a 35-cm-deep chisel plough every year. The manure application consisted of dry chicken pellets and dried cow manure during the alfalfa crop. All organic fields were irrigated using furrow irrigation except during the alfalfa crop when flood irrigation was used. The conventional field was originally under cotton–alfalfa rotation and continuous cotton during the last 6 years, and was furrow-irrigated and ploughed similar to the organically managed fields. Liquid fertilizer was applied at the preplant stage (10–34–0) followed by two applications of urea ammonium nitrate solution. However, the contribution of the current farming practices to the SIC stock and its nature were unclear. Further, this was an on-farm study and the interpretation of the data was challenging.

For example, it was difficult to quantitatively document material inputs to study units and to ascertain the similarity of baseline soil characteristics among sites (Jacinthe et al., 2011).

In North Dakota, USA, SIC stocks to 30.5-cm depth were about 3 times higher in a field under conventional practices than those after 19 years under OA practices (29.5 vs 9.8 Mg C ha⁻¹; Liebig and Doran, 1999). Crop rotation in the conventional field was spring wheat (*Triticum aestivum* L.)–sunflower (*Helianthus annuus*) vis-à-vis that organically managed field included oats (*Avena sativa* L.)–sweet clover (*Melilotus officinalis* L.)–rye (*Secale cereale* L.)–sunflower–buckwheat (*Fagopyrum esculentum*)–alfalfa (*M. sativa* L.)–spring wheat–flax (*Linum usitatissimum* L.)–pearl millet (*Pennisetum glaucum* L.). No data were available on the type and rate of fertilization of the conventional field. However, the organically managed field received composted cattle manure prior to seeding spring wheat at the rate of 5.6 Mg ha⁻¹. Differences in SIC stocks between the conventionally and organically managed fields can be explained by different rates of erosion, with rates being slower on the organic farm due to less frequent tillage and inclusion of cover crops in the cropping sequence. However, no differences were observed in SIC stocks to 30.5-cm depth among similar conventional and organic farms in North Dakota, and also after 9, 10, and 29 years of OA at farms in Nebraska, USA (Liebig and Doran, 1999).

The SIC stocks to 15-cm depth of a cropland cultivated with soybean (*Glycine max* L.) in Minnesota, USA, and managed organically for 5 years was similar to that of a conventionally managed field (21.2 and 24.2 Mg C ha⁻¹; calculated as the difference between total C and organic C; Phillips, 2007). Both fields were tilled to 15-cm depth and the organically managed field received manure whereas the conventional field was fertilized with urea (0.020 Mg N ha⁻¹, 0.052 Mg P ha⁻¹) and urea + potash (0.179 Mg N ha⁻¹, 0.067 Mg P ha⁻¹, 0.067 Mg K ha⁻¹; Phillips, 2007).

In a semiarid region in Spain, carbonate contents for 0–10, 10–20, and 20–30 cm depths of irrigated calcareous soils managed organically and farmed with wheat (*Triticum* spp.)–oats–peas (*Pisum sativum* L.) rotation for 18 years were lower (24.01, 24.43, and 26.61%, respectively), than those of conventionally managed and farmed with a wheat monoculture (32.69, 32.81, and 34.54%, respectively; Romanyà and Rovira, 2007). In contrast, after farming for 18 years, carbonate contents were comparable among organically and conventionally managed calcareous soils cultivated to barley (*Hordeum vulgare* L.)–fallow rotation system under rain fed conditions in the

same region (ranging between 28.18% and 30.14%). However, this study was conducted as a noncontrolled farm-level experiment making interpretations rather difficult (Romanyà and Rovira, 2007).

In conclusion, the knowledge about the effects of OA on SIC stocks is scanty and mainly based on observations in farmer's fields. However, it has been hypothesized that OA practices have the potential to alter SIC stocks. Thus, studies on C dynamics of soils under OA must include studies on SIC stocks, especially in arid and semiarid regions. Controlled research plot experiments are needed to determine the effects of OA practices including irrigation, liming, and addition of animal and green manures on SIC dynamics (Ahmad et al., 2015).

3.2 Soil Organic Carbon Stock

The effects of OA on SOC stocks have been studied more than those on SIC stocks. The rationale behind more emphasis on the former is that adequate SOC stock management is highly relevant to crop production in organic farming both from an agronomical and an ecological point of view (Brock et al., 2011). Specifically, the demand for fresh OM supply to maintain soil productivity is higher in OA than in conventional crop production systems because: (1) mineral fertilizer as an N source for crops and soil microorganisms is not applied in OA, (2) higher dependence on soil functions and SOC services in OA and, thus, demand for higher SOC stocks and turnover intensity, and (3) positive correlation between OM supply and turnover in soils (Leithold et al., 2015). Thus, it has been hypothesized that core practices of OA including returning plant residues and manures from livestock back to the land, and/or integrating perennial plants, mainly grass-clover mixtures, into the system reduces SOC losses, and either maintains SOC or even causes an increase in SOC stocks (Gattinger et al., 2012). For example, compared to conventional systems, a higher SOC stock replenishment due to crop rotations was reported for arable lands under OA in Austria (Kasper et al., 2015). However, organic yields are on average ~19% lower than those under conventional management, and this potentially results in lower direct plant-derived soil C inputs via rhizodeposition and decomposition of plant litter (Ponisio et al., 2015). Otherwise, the accumulation of SOC under OA practices despite fewer C inputs and greater soil tillage compared to conventional systems has been explained by more transformation of plant C into soil microbial biomass (Kallenbach et al., 2015). Thus, the efficiency and rate at which new C inputs are utilized by soil

microbes to build microbial biomass and subsequent necromass may be a potential mechanism for SOC accumulation under OA management.

3.2.1 Comparisons of Conventional and Organic Farming Systems

3.2.1.1 Metaanalyses

[Gattinger et al. \(2012\)](#) reported that SOC stocks of organically managed top soils (0–15 cm) were on average 3.50 Mg C ha⁻¹ higher than those under conventional management based on a metaanalysis of a large database comparing SOC in organic versus nonorganic farming systems. If only the highest quality data (ie, those based on measured soil bulk densities, and on measured external C and N inputs) were considered, the increase in SOC stock under organic management was reduced to 1.98 Mg C ha⁻¹. Apparently, differences in external C inputs and crop rotations were important for the higher SOC stocks under OA. The median age of the farming system comparisons was 10 years, and the median soil sampling depth was 0–15 cm. However, less than 50% of the data were of high quality (ie, measured soil bulk density and C and N inputs). Other limitations were the often missing baseline data on SOC stocks from the initiation of the conventional—organic farming systems comparisons ([Gattinger et al., 2012](#)). [Olson et al. \(2014\)](#) highlighted that pretreatment baseline of soil data are essential for proper field experimental design. Thus, it was not clear whether differences in SOC stocks among conventionally and organically managed plots already existed prior to the start of some of the experiments ([Gattinger et al., 2012](#)). Another limitation was the poor global coverage on geographical distribution since most of the data were reported from Australia, Europe, New Zealand, and North America. Finally, the shallow sampling depth was not sufficient to compare the effects of farming system on SOC as organic systems integrate perennial plants, mainly grass–clover mixtures that deposit SOC stocks at deeper soil depths than those generally studied ([Gattinger et al., 2012](#)). Some studies, especially those including subsoil depths, from tropical regions and from long-term experiments published since this metaanalysis was performed, will be discussed in the following section.

A recent metaanalysis comparing conventional and organic farming systems under Mediterranean croplands indicated that SOC sequestration rate in top soil (average soil depth 19.4 cm) increased by 0.97 Mg C ha⁻¹ yr⁻¹ under OA compared to those under conventional management ([Aguilera et al., 2013](#)). The SOC increment under OA was greater under

irrigation than under rainfed conditions (25 vs 13% increase over conventional, respectively). Further, a higher SOC sequestration gain under OA was achieved with compost than with raw manure, probably as compost contains more stabilized forms of C. The degree of intensification in C input rate was the main driver behind the SOC accumulation under OA. Nonetheless, it is not known whether the input of biomass-C brought in from outside the land unit was accounted for or not. This is an important methodological consideration (Olson et al., 2014). However, physical, economic, and social constraints contributed to lower application rates of OM on farms compared to those for the plot experiments. Thus, the best OA practices under Mediterranean conditions are not widely used at organic farms (Aguilera et al., 2013). This metaanalysis was, however, also limited by shallow soil sampling depth, missing baseline data, and missing data on soil bulk density. Further, SOC stock calculation was biased by changes in bulk density, and the short experiment duration (mostly between 3 and 10 years) also contributed to data uncertainty (Aguilera et al., 2013).

3.2.1.2 Soil Profile Studies

The data from one of the first studies comparing subsoil SOC stocks for conventional and OA systems after 18 years indicated that latter soils had lower SOC stocks by 28 Mg C ha⁻¹ in 0–120-cm depth, and by 7 Mg C ha⁻¹ less in the 0–30-cm depth than conventionally managed soils (Fig. 1; Bell et al., 2012). Further, the OA systems had a higher proportion of the SOC stock to 120-cm depth in the surface 30 cm (46%) compared with those under conventional management (42%). Lower C inputs in the OA systems may have contributed to lower leaching losses as dissolved organic carbon (DOC) and, thus, to a more shallow SOC stock distribution compared to the conventional systems (Bajgai et al., 2014). Both agricultural systems were under annual crop and alfalfa/crop rotations. However, the OA systems neither received manure nor compost. This, together with lower yields, may have contributed to lower SOC stocks compared to the conventional systems. Specifically, over the experimental period of 18 years there was a positive relationship between total C inputs and SOC stocks for all systems, that is, each 1 Mg C ha⁻¹ input corresponded to 0.15, 0.36, and 0.60 Mg C ha⁻¹ increase in SOC stock in 0–30, 0–60, and 0–120-cm depth, respectively (Bell et al., 2012).

Profile SOC stocks for a range of conventional and organic land-use systems have been studied in the Los Pedroches Valley, southern Spain.

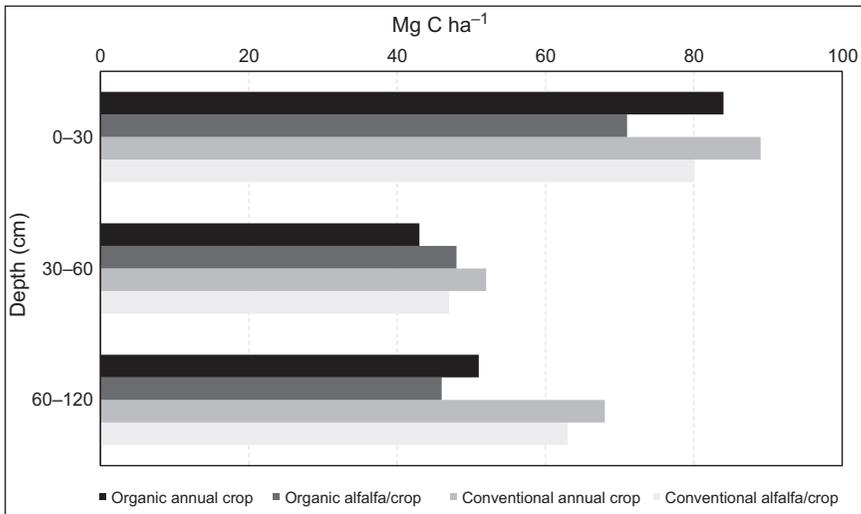


Figure 1 Soil organic carbon stocks (Mg C ha^{-1}) to 120-cm depth 18 years after establishing farming systems in Manitoba, Canada (Bell et al., 2012).

The SOC stock to 76.1-cm depth was $73.6 \text{ Mg C ha}^{-1}$ under olive (*Olea europaea* L.) groves managed by OA practices for ~ 20 years compared with $54.2 \text{ Mg C ha}^{-1}$ to 54.3-cm depth for conventionally managed olive groves (Lozano-García and Parras-Alcántara, 2013). The reduced tillage intensity under OA compared to conventionally managed and more intensively tilled olive groves may have partly contributed to this difference. However, soil profile SOC stocks of dehesas (Mediterranean grassland ecosystem with scattered oak trees—grazing system with *Quercus ilex* spp. *ballota*) for two soil types managed for 20 years by OA including no-till (NT) were not different from those of conventionally managed and tilled soils (76.4 and $43.3 \text{ Mg C ha}^{-1}$ vs 74.9 and $44.8 \text{ Mg C ha}^{-1}$ for Cambisols and Leptosols, respectively; Parras-Alcántara et al., 2014). Differences in climate, soil conditions, soil erosion rates, grazing systems, and water and nutrient management may have contributed to differences in SOC stocks. Soil profile SOC stocks were also studied for Cambisols, Luvisols, and Leptosols managed for 20 years under conventional and organic annual cereal–fallow rotation with the cereal types durum wheat (*Triticum durum*) or barley (*Hordeum vulgare* L.) (Fig. 2; Parras-Alcántara et al., 2015). The change from conventional to OA management resulted in higher soil profile SOC stocks. Supposedly, crop residues deposited on the soil surface decomposing slowly as a result of drier conditions and reduced mineral nutrient availability were not transferred to deeper

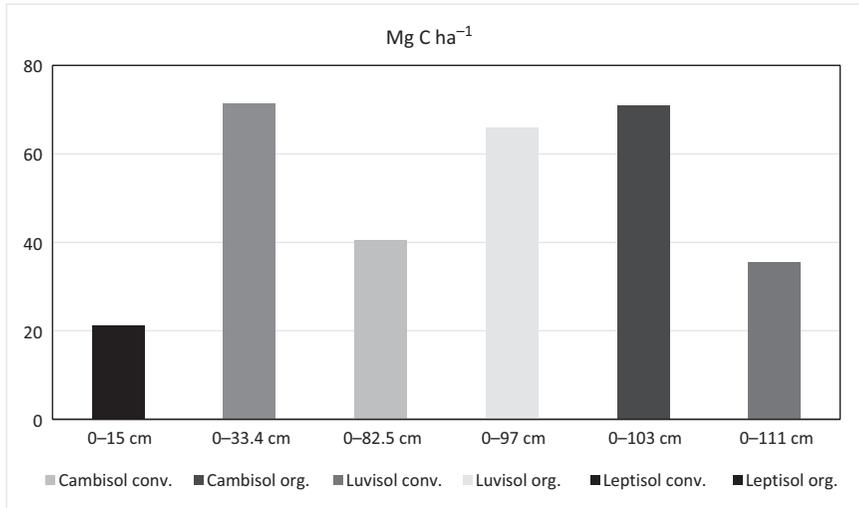


Figure 2 Profile soil organic carbon stocks (Mg C ha⁻¹) for Cambisols, Luvisols, and Leptosols after 20 years of farming systems management in southern Spain (Parras-Alcántara et al., 2015).

soil layers to the same extent under OA practices than those at the conventionally tilled plots. Specifically, SOC stocks to 82.5 and 103.0-cm depths at conventionally and OA managed Cambisols were 40.5 and 71.0 Mg C ha⁻¹, respectively. To 111.0 and 97.0-cm depths, Luvisols stored 35.5 and 66.0 Mg C ha⁻¹ under conventional and organic management, respectively. Further, Leptosols had SOC stocks of 21.3 Mg C ha⁻¹ to 15.0-cm depth under conventional practices and of 71.4 Mg C ha⁻¹ to 33.4-cm depth under OA practices (Parras-Alcántara et al., 2015). In conclusion, studies in the Los Pedroches Valley have shown that it is necessary to study entire soil profiles to assess the effects of conversions from conventional to OA practices on SOC stocks (Parras-Alcántara and Lozano-García, 2014).

3.2.1.3 Long-Term Experiments

Data from long-term experiments are a prerequisite for reliable conclusions about the effects of OA practices on SOC stocks, and some results will be discussed later in the chapter. For example, mean SOC stock changes in Ap horizons at long-term experiments in Germany and Switzerland were highly variable (Brock et al., 2012). Specifically, values ranged between gains of 240 and 522 kg SOC ha⁻¹ y⁻¹ over 5 years after plot establishment at an experiment in Germany to between losses of 93 kg SOC ha⁻¹ y⁻¹ and gains of

109 kg SOC ha⁻¹ y⁻¹ over 27 years after establishment of an experiment in Switzerland. However, SOC sequestration rates for conventional and OA systems at both sites did not differ. In contrast, the data from an 11-year-old experiment in Germany showed that SOC stock changes in the Ap horizon at a mixed OA site with cattle were higher than those at a conventional cash crop farm without animals (1581 vs 216 kg SOC ha⁻¹ y⁻¹, respectively). Otherwise, Ap horizons in a mixed OA farm with cattle at a 10-year-old experiment in Germany lost more SOC (ie, 1,947 kg SOC ha⁻¹ y⁻¹) than those of a mixed conventional farm with cattle (514 kg SOC ha⁻¹ y⁻¹). Brock et al. (2012) concluded that the impact of agricultural practices on SOC stocks is not an intrinsic characteristic of any farming system, but rather the result of the actual structure of the farming system, in particular, of the composition and management of crop rotations, and the availability and utilization of organic manure.

At a long-term experiment in Tuscany, Italy, soils farmed to crops under OA management for 15 years had higher SOC stocks to 30-cm depth than those under conventional management (27.9 vs 24.5 Mg C ha⁻¹; Lazzarini et al., 2014). Both systems did not differ significantly in SOC sequestration rates, with gains of 0.48 Mg SOC ha⁻¹ y⁻¹ for the organic and losses of 0.54 Mg SOC ha⁻¹ y⁻¹ for the conventional system, respectively. However, the SOC sequestration rates were based on analyses over only 4 years and, thus, probably over a too short time period. Also, bulk densities were not measured but estimated adding to uncertainties in the conclusions regarding the effects of farming practices on SOC stocks (Lazzarini et al., 2014).

Cavigelli et al. (2013) summarized the results from several long-term agricultural research sites (LTARs) in the United States including comparisons of OA with conventional NT systems. Differences in SOC stocks were variable between LTARs depending on the level of inputs of biomass-C. For example, after 18 years SOC stocks to 20-cm depth were similar for a NT corn-soybean (56.0 Mg C ha⁻¹) and organic corn-soybean-wheat/red clover (57.6 Mg C ha⁻¹) rotation system in Wisconsin, USA. The C inputs were about 25% lower in the organic compared to the conventional NT system due to lower crop residue yields in the organic system, suggesting that the form or placement of C inputs may have impacted the SOC stocks. Otherwise, after 11 years, SOC stocks to 1-m depth in corn-rye-soybean-wheat/legume rotations was 11% more in a manure-based OA (60.8 Mg C ha⁻¹) than in soil under a conventional NT system (54.9 Mg C ha⁻¹). Inputs of biomass-C to the soil were more in the OA than those under the

NT system, largely due to manure and/or compost additions. The results indicated that tilling under sufficient quantities of organic materials, particularly manure, into soil may be a more effective means of increasing SOC stocks than merely eliminating tillage (Cavigelli et al., 2013).

3.2.1.4 Organic Fertilizers

Input of biomass-C with a low C:N ratio applied as commercial organic fertilizers in an organic crop farm in the southern Piemonte region, Italy, in which green manuring was also used, were not efficient in enhancing SOC (Sacco et al., 2015). Specifically, SOC stocks to 35-cm depth 6 years after conversion to OA were reduced by 12.7 Mg C ha⁻¹ compared with reduction of 7.5 Mg C ha⁻¹ in an organic livestock farm cropping system in which nutrients were supplied from farmyard manure, and lower by 16.3 Mg C ha⁻¹ under conventional practices. However, the differences were only significant between the OA system receiving farmyard manure and the conventional system. Thus, farmyard manure better contained the depletion in SOC stocks (Sacco et al., 2015).

3.2.1.5 Tropical Regions

After >7 years under OA practices, SOC stocks to 25-cm depth for coffee (*Coffea arabica* L.) agroforestry systems in Costa Rica tended to be higher than those under conventional practices (73.0 vs 53.1 Mg C ha⁻¹, Häger, 2012). This trend may be due to more trees per hectare at the OA systems which significantly differed in stature and average wood density from those at the conventional farms. Further, organic management relied on soil improvement by incorporating vegetation elements, the application of organic amendments, green manure, and erosion barriers. However, baseline data on SOC stocks at the time of the conversion to OA management were not recorded. Also, no detailed information on soil texture was available and neither was the information on the depth distribution of SOC stocks. These factors might confound elucidating the true effect of farm type on the SOC storage (Häger, 2012).

3.2.2 Comparisons of Organic Farming Systems

3.2.2.1 Long-Term Experiments

After “organic” management for 50 years, SOC stocks to 40-cm depth at two long-term fertility experiments in Sweden were lower without (62.1 and

85.4 Mg C ha⁻¹ at Fors and Őrja, respectively) than with farmyard manure fertilization (66.2 and 88.9 Mg C ha⁻¹ at Fors and Őrja, respectively; Kirchmann et al., 2013). Soils without mineral N fertilization also had lower SOC stocks than those with mineral N fertilization (51.0 and 68.2 Mg C ha⁻¹ at Fors and Őrja vs 60.4 and 85.4 Mg C ha⁻¹ at Fors and Őrja, respectively). Thus, less C input through crop residues from low-yield treatments (ie, organic systems without N fertilization) provided less inputs for SOC formation (Kirchmann et al., 2013).

The SOC stocks to 15-cm depth of the winter wheat phase of arable crop rotations did not differ between farms that have been under conventional or organic management in Austria since the last 16 years (van Leeuwen et al., 2015). Specifically, SOC stocks were 28.0 and 25.7 Mg C ha⁻¹ for the conventional and organic systems, respectively.

3.2.2.2 Livestock Husbandry and Animal Manure

Livestock husbandry may be important in reducing the detrimental effects of arable OA practices on SOC stocks. For example, after 11 years under OA, SOC stocks to 30-cm depth at a crop farm with cash crops in Germany were lower than those at a mixed farm with animal husbandry (47.9 vs 54.1 Mg C ha⁻¹), but not different from those at a farm with rotational ley without animals (51.3 Mg C ha⁻¹; Schulz et al., 2014). However, SOC stocks at 30–60 and 60–90-cm depths were comparable among OA practices as the duration of the experiments was probably not long enough to alter SOC stocks in the subsoil (Schulz et al., 2014).

3.2.2.3 Soil Tillage and Green Manure

Reducing and/or eliminating tillage may minimize the detrimental effects of OA practices on SOC stocks. For example, SOC stocks to 10-cm depth in >5-year-old organic olive farms in Spain were lower when managed by tillage compared to those managed by mowing (18.6 vs 59.3 Mg C ha⁻¹; Soriano et al., 2014). In comparison, the SOC stock in 0–10-cm depth at nearby natural areas was 67.8 Mg C ha⁻¹. Further, the OA practices with tillage or mowing had lower SOC stocks at 10–20-cm depth compared to those under natural areas (13.6 and 18.6 Mg C ha⁻¹ vs 36.8 Mg C ha⁻¹, respectively; Soriano et al., 2014). Otherwise, reduced tillage, and reduced tillage with green manure at organic rainfed almond (*Prunusdulcis* Mill) farms in the Murcia Region (southeast Spain) led to an increase of ~48% in the SOC stock to 15-cm depth after 4 years of establishment at one of the sites (Almagro et al., 2013). At another site, the incorporation of green manure

resulted in a significant increase of $\sim 26\%$ in the SOC stock to 15-cm depth compared to that under reduced tillage without green manure.

Besides reducing tillage depth, another approach to reduce detrimental tillage effects of OA practices on SOC stocks may be changing the mode or type of tillage. For example, in a 12-year trial at a mixed organic farm in Germany with a typical crop rotation, SOC stocks to 60-cm depth were 75, 83, 91, and 93 Mg C ha⁻¹ under double-layer plow, deep moldboard plow, shallow moldboard plow, and chisel plow, respectively (Zikeli et al., 2013).

Positive effects on SOC stocks by replacing conventional tillage (CT) in organic vegetable farming with NT and weed cover mulching were reported by Yagioka et al. (2015). Specifically, during the first 3 years after the establishment of the farming practice, the rate of change of SOC to 30-cm depth under CT were -4.14 and -3.13 Mg C ha⁻¹ y⁻¹ depending on N fertilization levels compared with those of 0.17 and 0.39 Mg C ha⁻¹ y⁻¹ under NT along with weed cover mulching (Yagioka et al., 2015).

3.2.2.4 Organic Fertilizers

The choice of organic fertilizers is an important issue in organic vegetable production. For example, adding similar amounts of compost-C to organic lettuce (*Lactucasativa* var. *longifolia* Lam. cv. Bacio) by applying compost made from olive pomace mixtures either stopped at the active phase or processed until maturation resulted in a loss of 0.20 Mg SOC ha⁻¹ or an increase by 3.04 Mg SOC ha⁻¹ to 30-cm depth over 3 years (Montemurro et al., 2015). In contrast to amendments with high C:N ratios, the use of mature compost from a mixture with a low C:N ratio appeared to be the most suitable for organic lettuce production.

In conclusion, SOC stocks in the surface soil can be higher under OA than those under conventional practices, but changes in subsoil SOC stocks are not widely studied. Specifically, the effects of OA practices on SOC stocks depend on climate, soil conditions, soil erosion rates, grazing systems, and water and nutrient management. Since yields under OA systems are lower, direct soil C inputs are also lower compared to those under conventional systems. Thus, crop rotations along with the use of legume-grass leys, manure application, and animal husbandry are important practices of reducing SOC losses under OA. However, properly designed long-term experiments including baseline data on profile SOC stocks for the conventional-organic systems comparisons, measurements of soil bulk density, and equivalent soil mass calculations are needed to credibly assess the effects of OA on SOC stocks.



4. SOIL-DERIVED GREENHOUSE GAS FLUXES UNDER ORGANIC AGRICULTURE

Agricultural systems emit GHGs from: (1) fossil fuel use in machinery, (2) enteric fermentation, (3) the management of agricultural soils, (4) manure deposited on pasture, (5) synthetic fertilizers, (6) rice (*Oryza sativa* L.) cultivation, (7) manure management, (8) crop residues, (9) biomass burning, and (10) manure application (Smith et al., 2014). Specifically, organic and inorganic materials applied to agricultural soils are decomposed through biotic processes, releasing significant amounts of CO₂, CH₄, and nitrous oxide (N₂O) to the atmosphere. However, CO₂ emissions or uptake from agricultural SOC management are only a small portion of the total forest and other land use (FOLU) emissions, and, thus, are not reported to United Nations Framework Convention on Climate Change (UNFCCC) under current climate agreements, and are typically not included in regional or global GHG estimates (Tubiello et al., 2013). Otherwise, annual total non-CO₂ GHG emissions from agriculture in 2010 were estimated at 5.2–5.8 GtCO₂eq y⁻¹, comprising about 10–12% of global anthropogenic emissions (Smith et al., 2014). The enteric fermentation and agricultural soils represented together about 70% of total emissions, followed by paddy rice cultivation (9–11%), biomass burning (6–12%), and manure management (7–8%). Paddy rice cultivation was a major source of global CH₄ emissions, which in 2010 were estimated at 493–723 MtCO₂eq y⁻¹ (Smith et al., 2014). From 2000 to 2010, cattle contributed the largest share (ie, 75% of the total emissions from enteric fermentation), followed by buffalo, sheep, and goats (FAOSTAT, 2013). Manure deposited on pastures led to far larger emissions than that applied to soils. Further, two-thirds of the total manure emissions came from grazing cattle, with smaller contributions from sheep and goats. However, considering current trends, synthetic fertilizers will become a larger source of GHG emissions in less than 10 years than manure deposited on pasture, and it will be the second largest of all agricultural emission categories after enteric fermentation (Smith et al., 2014).

It is often assumed that OA is associated with lower levels of GHG emissions (McGee, 2015). However, this assumption cannot be generalized as crop yields under OA can be lower than those under conventional systems, on-farm energy use can be higher on organic farms, and production and delivery of large quantities of organic fertilizer can contribute to high GHG emissions in some organic systems. Further, higher GHG emissions in OA

may not be offset by those of conventional farming associated with the manufacture of synthetic fertilizers and pesticides. McGee (2015) reported that increase in certified organic farming in the United States is increasing both the total amount of GHG emitted from agricultural production and the intensity of GHGs emitted per hectare of agricultural land. Thus, OA practices applied at the scale of conventional agricultural production may emit more GHG than conventional farming due to lower agronomic yields and heavy reliance on machinery (McGee, 2015).

Structural variables affecting GHG emissions for conventional and OA systems have been compared by metaanalysis (Lee et al., 2015). In about two-thirds of 195 observations, OA had lower GHG emissions than conventional farming. Further, OA was superior to conventional farming regarding GHG emissions for field crops, dairy, and mixed crop farms. Contrarily, OA was less likely to be superior in GHG emissions for livestock, vegetable, and fruit farms. However, superior GHG emission effects for OA were highly dependent on the unit or basis of measurement. Output-based (ratio/Mg) measures significantly reduced the superiority of GHG emissions effects for OA in comparison to area-based (ratio/ha) measures due to yield differences. Among limitations of this metaanalysis was the narrow geographical distribution because most studies were from Europe, and did not consider nutrient spillover effects in conventional–organic conversions (Lee et al., 2015).

The magnitude of soil-derived GHG fluxes affected by OA are discussed in the following section. These practices include the avoidance of synthetic fertilizers, management of grazing animals, and animal manure, crop residue and green manure management, and soil tillage for controlling weeds and incorporation of manure and crop residues.

4.1 Carbon Dioxide

Published data on direct measurements of CO₂ emissions from paired conventional and OA experiments are scanty. Indirectly, the contribution of OA practices to the atmospheric CO₂ concentration can be estimated by assessing changes in soil C stocks. For example, a metaanalysis by Gattinger et al. (2012) reported the maximum SOC sequestration potential of 1.65 Mg CO₂ ha⁻¹ y⁻¹ in the top 20-cm layer by the adoption of OA practices. However, rates of SOC sequestration were uncertain because only arable and vegetable land-use types were compared, only 6 out of 8 climate zones were considered, and no data from Africa were available (Gattinger et al., 2012). There were also uncertainties in data about additional 3.56 Mg CO₂ ha⁻¹ y⁻¹ sequestered in

some soils of Mediterranean OA farming systems compared to those under conventional management (Aguilera et al., 2013). Some studies reporting direct CO₂ emissions from agricultural soils for organic–conventional comparisons, and from studies comparing different OA practices will be discussed later.

4.1.1 Comparisons of Conventional and Organic Farming Systems

Kontopoulou et al. (2015) monitored CO₂ emissions under irrigated conventional and irrigated OA management of common bean (*Phaseolus vulgaris* cv. “contender”) under Mediterranean climate. The cumulative CO₂ emissions in the 84-days cropping period were higher under OA (2.5 and 2.8 Mg CO₂–C ha⁻¹ for high and low-salinity irrigation water, respectively) than those under conventional management (2.1 and 2.3 Mg CO₂ ha⁻¹ for high- and low-salinity irrigation water, respectively). The higher rates of CO₂ emission in the OA may have arisen from respiration of added compost. Application of compost may have also improved: (1) soil structure and the continuity of pore space, (2) root penetration and flow of water and gases, thereby promoting OM decomposition, and (3) root exudation and, thus, microbial activity which may have enhanced microbial respiration in the rhizosphere (Kontopoulou et al., 2015).

Five years after establishing conventional and OA fruit production systems in Belgium, the soil CO₂ efflux measured over four short periods between May and Oct. was higher in the OA orchard compared to the conventionally managed orchard (Jamar et al., 2010). However, data variability was also high.

4.1.2 Comparisons of Organic Farming Systems

Within the first 3 years after conversion to OA, annual CO₂ emissions from OA rotations including pumpkin (*Cucurbita* spp.) as the main crop, and mixed cropping of okra (*Abelmoschus esculentus* L.), bell pepper (*Capsicum annuum* L.), and eggplant (*Solanum melongena* L.) did not differ from that under control without fertilization and the treatment receiving organic amendments (Yagioka et al., 2015). Among OA systems, annual emissions in NT with weed cover mulching and conventional till were 4.43 and 3.99 Mg CO₂–C ha⁻¹ y⁻¹, respectively in 2011 compared with 4.24 and 3.26 Mg CO₂–C ha⁻¹ y⁻¹ in NT with weed cover mulching and conventional till in 2012, respectively (Yagioka et al., 2015).

Growing-season CO₂ evolution rates did not differ among semileafless field peas (*P. sativum* L. var. Santana KWS, Einbeck) and oats (*A. sativa* L. var. Dominik) grown under OA as sole crops or as intercrops, and were also not

affected by three fertilizer treatments (Jannoura et al., 2014). However, CO₂ effluxes among fertilizer treatments, averaged over all cropping systems, were different at 4.8, 9.4, and 6.4 Mg CO₂-C ha⁻¹ for 133 days from the control, horse manure, and yard-waste compost treatments, respectively. Higher proportions of more readily decomposable biomass-C in horse manure may have contributed to the high CO₂ emissions for this fertilizer treatment (Jannoura et al., 2014).

The annual CO₂ emissions from OA rainfed almond farms under reduced tillage with green manure, reduced tillage, and NT were similar at 518, 495, and 465 g C m⁻² y⁻¹, respectively (Almagro et al., 2013). Green manuring was expected to enhance CO₂ emissions because of higher C and N inputs, and increased microbial biomass. However, the formation of aggregates and their stabilization may also have been enhanced by green manuring along with increase in physical protection of SOC, thereby decreasing CO₂ emissions compared to soils under reduced tillage without green manure (Almagro et al., 2013).

In Denmark, Vinther et al. (2004) studied the impacts of crop rotations and input of OM in the form of green manure crops, straw residues, and incorporation of catch crops on soil respiration in unfertilized crop rotations with varying input of plant residues. Specifically, high-input rotations with a grass-clover crop and catch crops included were compared to low-input cereal rotations without catch crops. Soil respiration during the growing period varied considerably 4 years after the rotations were established. However, soil respiration did not differ between high- and low-input rotations with values ranging between 4.3 and 5.4 Mg CO₂-C ha⁻¹ during the growing period (Vinther et al., 2004).

At another site in Denmark, soil CO₂ emissions were studied during several monitoring periods for an OA rotation including barley undersown with grass-clover, 2 years of grass-clover and winter wheat with the N₂-fixing grass-clover mixture used as green manure by soil incorporation before sowing of the subsequent maize crop (Carter et al., 2012). The cumulative soil CO₂ emissions obtained by linear interpolation between measurements increased in the order unfertilized < digested slurry + maize < raw slurry < green manure. Specifically, emissions from the unfertilized plots were 259 and 235 g C m⁻² in 2007 and 2008, respectively, and 445 and 444 g C m⁻² in 2007 and 2008, respectively, for the plots receiving green manure. However, although green manure gave rise to the highest soil CO₂ emission, yet this treatment led to the highest near-term C sequestration potential among the three treatments (Carter et al., 2012).

4.2 Nitrous Oxide

The generally lower N input level for soils under OA compared to those under conventional management practices supports the expectation of lower soil N₂O emissions (Muller and Aubert, 2014). However, scientific evidence in support of this perception is scanty. Skinner et al. (2014) performed the first systematic literature review of pair-wise comparisons of organic and conventional farming systems followed by a metaanalysis of reports covering at least one annual measurement period. Soils under OA emitted N₂O at rates of 492 kg CO₂ eq. ha⁻¹ y⁻¹, that is, at lower rates than those managed conventionally. This corresponded to saving of 1.05 kg N ha⁻¹ y⁻¹ due to less N loss in form of N₂O emissions. The emissions reduction under OA practices may have been mainly due to significant reductions under arable cropping (497 kg CO₂ eq. ha⁻¹ y⁻¹) corresponding to 1.06 kg N ha⁻¹ y⁻¹, but comparative studies for grasslands and rice paddies were scanty. Lower N inputs were applied to organically managed soils, and N sources (ie, organic fertilizers and legumes) were less available compared to soils managed by conventional practices (Muller and Aubert, 2014). However, in the metaanalysis, no relationship between N inputs and N₂O emissions was observed for OA practices. Thus, Skinner et al. (2014) hypothesized that due to the delayed release of mineral N from organic sources a substantial part of the resulting N₂O emissions may become effective later than the vegetation period under study. Another possible explanation for the missing relationship may be the levels of background emissions, that is, the N₂O release from the mineralization of OM which may exceed the N₂O release by N input from the present year. Scaled to crop yields, the metaanalysis revealed 42.4 kg CO₂ eq. Mg⁻¹ dry matter (DM) more N₂O emitted from organically managed soils, but the database was rather weak. The yield gap between OA and conventional farming management was 26%. In conclusion, some evidence was available for lower N₂O emissions from OA managed soils when scaled to the area of cultivated land but higher emissions when crop yield-scaled (Skinner et al., 2014). In the following section some comparative studies not included in the metaanalysis, and comparisons of different OA practices will be discussed.

4.2.1 Comparisons of Conventional and Organic Farming Systems

4.2.1.1 Metaanalyses

A metaanalysis of studies, mainly from Europe, was limited as many studies reported only single measurement without standard deviations

(Mondelaers et al., 2009). Accordingly, N_2O emissions per unit area were lower under OA compared to those under conventional practices. By extending the database and including only studies from Europe, Tuomisto et al. (2012) reported that median N_2O emissions were 31% lower from OA systems per unit of field area mainly because of lower N inputs compared to that under conventional systems. Otherwise, median N_2O emissions under OA were 8% higher per unit of product (Tuomisto et al., 2012).

4.2.1.2 Organic Fertilizers

In a perennial apple (*Malus domestica* Borkh., 1803) orchard, soil N_2O emissions from OA plots fertilized either with composted chicken manure or alfalfa meal did not differ in the month following fertilizer application from those of conventional plots fertilized with $\text{Ca}(\text{NO}_3)_2$ (Kramer et al., 2006). However, denitrification efficiency under OA was enhanced probably because of: (1) increased C inputs from grass roots and fertilizer; (2) higher SOC and N contents; (3) larger, more active microbial communities; and (4) differences in the functioning of the denitrifier communities (Kramer et al., 2006).

Cumulative N_2O emissions during winter at plots cultivated with soybean were higher for manure-amended organic plots ($1.63 \text{ kg N}_2\text{O-N ha}^{-1}$) than those from unamended and conventionally managed plots ($0.64 \text{ kg N}_2\text{O-N ha}^{-1}$; Phillips, 2007). However, more studies are needed on interactions between timing of application of organic amendment and N_2O emissions for developing strategies for optimum N conservation in OA. Further, as large emission spikes can occur on short time frames, appropriate sampling strategies are prerequisite for calculating annual N_2O emissions (van der Weerden et al., 2000; Smukler et al., 2012).

4.2.2 Comparisons of Organic Farming Systems

4.2.2.1 Organic Fertilizers

Seasonal N_2O emissions of grazing land and cropland (without manure) organic arable farming rotations were measured between 2006 and 2009 during relatively wet seasons in eastern Scotland (Ball et al., 2014). There was appreciable variability in fluxes of N_2O measured across sites and seasons. The N_2O emissions from the arable land (1.9 and $3.0 \text{ kg N}_2\text{O-N ha}^{-1}$ in 2006 and 2007, respectively) exceeded those from the grass-clover (0.8 and $1.1 \text{ kg N}_2\text{O-N ha}^{-1}$ in 2006 and 2007, respectively). However, wet weather delayed manure applications in 2008 and emissions from the grass-clover

increased to 2.8 kg N₂O–N ha⁻¹. Nevertheless, organic grassland provided the most effective overall mitigation of N₂O emissions (Ball et al., 2014).

The effects of grass–clover management on N₂O emissions were studied for about 1 year in an organic arable land rotation on a sandy loam soil in a cool temperate climate (Brozyna et al., 2013). Mean annual N₂O emissions including all crop rotations did not vary among manure treatments. However, emissions were higher for spring barley (ie, 1.4 kg N₂O–N ha⁻¹ y⁻¹) after plant material from grass–clover cuts was left in the field to decompose and no fertilizer or manure was applied to any crop in the rotation compared to those when plant material from grass–clover cuts was harvested and equivalent amounts of N in digested manure were used for fertilization of food crops in the rotation [spring barley, potato (*Solanum tuberosum* L.) and winter wheat; 0.9 kg N₂O–N ha⁻¹ y⁻¹]. Further, large N₂O emissions were obtained after spring incorporation of cover crop and grass–clover residues as recently incorporated N-rich residues of legume-rich cover crops and grass–clover appear to have been the main driver of N₂O emissions. Cover crops and grass–clover are normally promoted to increase soil fertility and crop N supply in OA. However, they apparently also considerably increase the risk of N₂O emissions and may, thus, from a GHG perspective, counteract the positive effects of cover crops and green manure crops in maintaining SOC stocks. Thus, there is a need to further study how N₂O emissions can be reduced in organic farming systems (Brozyna et al., 2013).

Carter et al. (2012) studied the effects of the organically managed biofuel feedstocks, such as, dried straw of sole cropped rye, sole cropped vetch (*Vicia villosa* Roth cv. Latiga), and intercropped rye–vetch, as well as fresh grass–clover and whole crop maize on N₂O emissions during two measurement periods within 2 years. When pooling the N₂O emissions from the rye crop in the two periods, emissions from the plots fertilized with raw cattle slurry or a mixture of digested slurry and maize were similar to those at the unfertilized control (368 and 388 mg N m⁻², respectively). Concerning total N₂O emissions from maize plots, application of green manure more than doubled the emissions (235 and 277 mg N m⁻² in 2008 and 2009, respectively) compared to the unfertilized control (106 and 88 mg N m⁻² in 2008 and 2009, respectively). An even further increase occurred in plots fertilized with either raw slurry (total emissions of 641 and 670 mg N m⁻² in 2008 and 2009, respectively) or digested slurry + maize (total emissions of 943 and 444 mg N m⁻² in 2008 and 2009, respectively). Among the unfertilized plots, the largest loss of N₂O occurred from the vetch plots

(cumulative emissions per period between 45 and 719 mg N m⁻²). This risk of high N₂O emissions from sole cropping of vetch should be avoided (Carter et al., 2012).

Nadeem et al. (2012) monitored N₂O emissions of an ungrazed organic cereal production system consisting of grass–clover undersown in spring barley, and a full-year grass–clover ley followed by a spring barley crop. In general, green manure ley (mulched or harvested) increased N₂O emissions relative to a cereal reference with low mineral N fertilization. Specifically, during the 204-days measurement period N₂O emissions decreased in the order green manure mulched (3.26 kg N₂O–N ha⁻¹) > green manure harvested (2.89 kg N₂O–N ha⁻¹) > cereal (2.41 kg N₂O–N ha⁻¹; Nadeem et al., 2012).

The N₂O emissions decreased by 38% for an arable organic ungrazed cropping system when crop residues and the clover–grass ley were harvested, digested, and the effluents reallocated within the same cropping system, in comparison to mulching and incorporation of the biomass as green manure (Möller and Stinner, 2009). Specifically, emissions as sum of the season between two successive summers were 2.9 kg N₂O–N ha⁻¹ for ungrazed management, and crop residue incorporated in the field including clover–grass ley and cover crops. Under ungrazed management with digestion of field residues and crop residues harvested (inclusive clover–grass ley), digested, and effluents reallocated as manure within crop rotation, N₂O emissions were 1.8 kg N₂O–N ha⁻¹. Further, injection of liquid cattle slurry resulted in a strong increase of N₂O emissions (Möller and Stinner, 2009).

Ball et al. (2007) studied how ploughing date of grass–clover leys within an organic ley–arable rotation and timing of cessation of grazing before ploughing affects N₂O losses of the first cereal crop. Cumulative N₂O emissions were the highest (~8 kg N₂O–N ha⁻¹ over 17 months) after cessation of grazing in spring before ploughing, and the lowest (~5.5 kg N₂O–N ha⁻¹) after cessation of grazing in winter before ploughing. As emissions increased with temperature and rainfall, Ball et al. (2007) recommended to restrict tillage operations to cool, dry conditions, and being aware of possible soil compaction which may also promote denitrification.

Within an organic ley–arable rotation including FYM application, N₂O fluxes were not different in the ley and arable phases and in organic permanent grass (*Lolium perenne* L.) ranging between 2.9 and 3.0 kg N₂O–N ha⁻¹ y⁻¹ throughout the 3-year phase duration (Ball et al., 2002). However, some N₂O losses from the arable component were relatively high, and seasonal rainfall had a major influence on cumulative emissions of N₂O.

4.2.2.2 Catch Crops

Li et al. (2015) studied the effects of legume-based and nonlegume-based catch crops (cover crops) on soil N_2O emissions in an organically managed spring barley system for 1 year. The annual N_2O emissions from all catch crop treatments were comparable ranging between $527 \text{ g N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ for perennial ryegrass and $815 \text{ g N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ for red clover-ryegrass when catch crops were harvested in fall. Further, emissions ranged between $509 \text{ g N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ for perennial ryegrass and $841 \text{ g N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ for red clover when catch crops were incorporated by ploughing in spring. The exception was fodder radish (*Raphanus sativus* L., cv. Lunetta) treatment which had the highest annual emissions (1714 and $1180 \text{ g N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ either harvested or incorporated, respectively). Fodder radish had also the highest yield-scaled emissions (635 and $363 \text{ g N}_2\text{O-N Mg}^{-1}$ harvested or incorporated, respectively), whereas legume-based catch crop treatments tended to have the lowest emissions (168 – $204 \text{ g N}_2\text{O-N Mg}^{-1}$). Li et al. (2015) concluded that in comparison with nonlegume-based catch crops, legume-based catch crops have the potential to partly replace the manure application in organic cropping systems without causing higher N_2O emissions.

4.2.2.3 Tillage

Tillage effects on gaseous emissions vary among farming systems, soil type, and climate. For example, in a mixed cropping system, tillage and fertilizer management had no effects on monthly N_2O emissions for fertilized and unfertilized NT with weed cover mulching and CT during a 3-year observation period (Yagioka et al., 2015). Further, annual N_2O fluxes ranged between -0.035 and $0.310 \text{ kg N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$. However, CT showed higher N_2O emissions after tillage, and a higher soil nitrate content to 30-cm depth may have contributed to the enhanced denitrification compared to NT (Yagioka et al., 2015).

4.3 Methane

The CH_4 sink function of drained or well-aerated soils is based on the activity of specific CH_4 - and ammonium oxidizing bacteria, and on site-specific conditions (Skinner et al., 2014). Specifically, CH_4 uptake is correlated negatively with soil moisture since it regulates the diffusion of atmospheric CH_4 into the soil. High mineral N contents in soil (ammonium and nitrate) suppress CH_4 uptake. Well-aerated agricultural soils can

also turn into CH₄ sources for a certain period of time if, for example, cattle manure rich in OM and rumen-derived methanogens is regularly applied or if the soil is strongly compacted. However, rice paddies and waterlogged anaerobic systems are emitters of large amounts of CH₄ produced by methanogenic *Archaea*. Rice is usually grown in flooded conditions for weed management, and under these conditions anaerobic microbial decomposition of OM and organic fertilizers leads to CH₄ emissions (Muller and Aubert, 2014).

The lower N input level in OA systems compared to nonorganic systems supports the hypothesis of enhanced CH₄ uptake in organically managed soils (Skinner et al., 2014). Thus, soils under organic management take-up more CH₄ than those under conventional management (-0.61 vs -0.54 kg CH₄-C ha⁻¹ y⁻¹), corresponding to GHG mitigation of 20.2 and 18.0 kg CO₂-eq. ha⁻¹ y⁻¹, respectively. The lower CH₄ uptake by conventionally managed soils may be explained by higher mineral N contents in the soil solution compared to those under OA management which may suppresses the activity of the relevant enzymes for microbial CH₄ oxidation. In contrast, CH₄ emissions from rice paddies are a strong CH₄ source under both management systems, accounting for 6023 kg CO₂ eq. ha⁻¹ y⁻¹ under organic and 4857 kg CO₂ eq. ha⁻¹ y⁻¹ under conventional management. However, this observation was based on only one study with three comparisons, and, apparently, organic fertilizer applied in the OA system favored CH₄ production from the anaerobic decay of OM in rice paddies whereas the conventional systems did not receive organic fertilizer (Skinner et al., 2014). Thus, CH₄ emissions from organic fertilizers are a particular challenge for OA (Muller and Aubert, 2014). However, because of a small number of studies covering annual measurements, farming systems impact on soil CH₄ fluxes in rice paddies and upland agricultural soils could not be differentiated by metaanalysis (Skinner et al., 2014). Studies published after the metaanalysis was performed, and those comparing different OA systems will be discussed in the following section.

4.3.1 Comparisons of Organic Farming Systems

4.3.1.1 Organic Fertilizers

Anaerobic digestion of cattle slurry and maize silage can reduce slurry-derived soil CH₄ emissions from organically managed fields used for biofuel feedstock production (Carter et al., 2012). Specifically, soils under maize crop amended with raw slurry lost 2.2 mg CH₄-C m⁻² whereas those

amended with a mixture of anaerobic digested slurry and maize oxidized $-0.2 \text{ mg CH}_4\text{-C m}^{-2}$. However, this positive effect was only observed in one of two study years. The CH_4 emissions closely followed the amount of C applied being a source for methanogenesis. However, CH_4 was a negligible GHG source compared to N_2O emission (Carter et al., 2012).

Annual CH_4 oxidation rates were $\sim 60\%$ higher under organic ungrazed management, crop residue incorporated in the field including clover–grass ley and cover crops ($1.158 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$), compared to harvesting crop residues, digesting clover–grass ley and reallocating the effluents within the same cropping system without ($0.441 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$), and with external inputs ($0.514 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$; Möller and Stinner, 2009). Further, the influence of crop type or manuring on soil CH_4 uptake was negligible. However, CH_4 oxidation was reduced after reallocation of effluents to spring wheat probably due to a higher N supply as it has been shown that NH_4^+ substantially reduces CH_4 oxidation (Möller and Stinner, 2009).

4.3.1.2 Tillage

Type of tillage and surface conditions can strongly impact methanogenesis. For example, in a mixed organic cropping system, CH_4 uptake over 3 years did not differ between NT with weed–cover mulching and CT (ranging between -0.108 and $-1.738 \text{ kg CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$; Yagioka et al., 2015). However, CH_4 uptake under NT tended to increase with time as OM inputs by weeds caused an increase in air-filled porosity and a decrease in bulk density both contributing to increased CH_4 uptake (Yagioka et al., 2015).

In conclusion, published direct measurements of soil CO_2 , N_2O , and CH_4 emissions from paired conventional and OA field experiments are scanty, and often performed only in temperate regions. Net CO_2 emissions from soils under OA may be lower than those for conventional practices but increased CO_2 efflux may occur from OA managed soil after addition of organic fertilizers. Some evidence was found for lower N_2O emissions from OA managed soils when scaled to the area of cultivated land but higher N_2O emissions when crop yield-scaled. However, there may be a trade-off for OA as higher SOC stocks may correlate with higher N_2O emissions (Muller and Aubert, 2014). Further, the net CH_4 emissions from OA managed soils may be lower than those from conventionally managed soils. However, it is unclear whether organic rice cultivation results in higher soil CH_4 emissions compared to conventional rice cultivation.



5. EFFECTS OF ORGANIC AGRICULTURE ON YIELD

Agronomic yields for OA are presumably lower than those achieved by conventional agricultural practices as soluble mineral inputs are prohibited, and synthetic herbicides and pesticides are prohibited in favor of natural pesticides (Trewavas, 2001). However, yield differences between organic and conventional practices are highly contextual, depending on system and site characteristics, but the factors limiting organic yields are not fully understood (Seufert et al., 2012). Thus, with increased experience and timely weed management, organic yield performance has been shown to improve at several long-term organic comparison trials in the United States (Delate et al., 2015). The addition of manure, along with legume forages/cover crops, in long-term organic fertility schemes has proven essential for sufficient soil quality to support optimal yields across sites. Recently, Ponisio et al. (2015) reported a more robust estimate of the gap between organic and conventional yields compared to previous metaanalyses. This approach was based on a hierarchical metaanalytic framework that overcame the methodological pitfalls of other studies by accounting for both the multilevel nature of the data and the yield variation within studies. Further, a more extensive and up-to-date metadataset was compiled, comprising over 3 times the number of observations of any of the previous analyses (Ponisio et al., 2015).

Ponisio et al. (2015) estimated that organic yields were on average 19.2% lower than conventional yields despite historically low rates of investment in organic cropping systems. However, the overrepresentation of specific practices or crops in the dataset may have excessively influenced and biased this estimate. For example, cereal crops, which exhibited the greatest difference in yield of the crop types between organic and conventional systems, were greatly overrepresented (53% of comparisons). The observation that cereal productivity (including wheat, barley, rice, and maize) is lower under OA is of interest because of the importance of cereals in the human diet and their predominance in cultivated land area. However, the large yield difference between organic and conventional farming systems for cereals is not surprising, given the extensive efforts since the Green Revolution to increase cereal yields by breeding high-yielding cereal varieties adapted to conventional inputs. Another limitation in the dataset was that many comparisons between OA and conventional agriculture use modern crop varieties selected for their ability to produce under high-input (conventional) systems. Such varieties

are known to lack important traits needed for productivity in low-input systems, potentially biasing toward finding lower yields in organic versus conventional systems. In contrast, few modern varieties have been developed to produce high yields under OA. The development of such breeds would be an important step toward reducing yield gaps. Another limitation was that apparently the available literature was favoring studies reporting higher conventional than OA yields, and, thus, the yield gap reported may have been an overestimation (Ponisio et al., 2015).

The yield ratios (OA yield/conventional yield) of most crop types did not vary among one another (Ponisio et al., 2015). However, yield ratios for apple (*M. domestica* Borkh., 1803), maize, oat, soybean, and tomato (*Solanum lycopersicum* L.) were higher than those for barley, potato, and wheat. Otherwise, no differences in yields for leguminous and nonleguminous crops nor for perennials and annuals were observed nor between the yield gaps for studies conducted in developed versus developing countries. When N inputs were similar between organic and conventional treatments, the yield gap was 9% but the yield gap was 30% when N inputs were greater in conventional treatments. When N inputs were higher in the OA treatments, the yield gap was intermediate (17%) and marginally different from the yield ratio with similar N input. Similarly, low-input conventional systems had a smaller yield gap than high-input systems (Ponisio et al., 2015).

Ponisio et al. (2015) concluded that the yield under OA may be improved by management practices that diversify crop fields in space or over time, that is, multicropping systems and crop rotations. This was based on the observation that the yield gap between organic polycultures and conventional monocultures (9%) was smaller than when both treatments were monocultures (17%) or both polycultures (21%). Similarly, the yield gap was smaller when the OA system had more rotations (8%) compared with when both treatments had a similar number of rotations (20%) or did not have crop rotations at all (16%). The results of this metaanalysis also suggested that polyculture and crop rotations increase yields in both OA and conventional cropping systems. Thus, additional investment in agroecological research has the potential to improve productivity under OA to equal or better than conventional yields in specific cropping systems (Ponisio et al., 2015). In the following section, some studies comparing yields of conventional and OA systems with the major cereals (corn, rice, wheat, and barley), and an oilseed (soybean) are discussed by using published data since the last search for the metadataset was conducted by Ponisio et al. (2015).

5.1 Tropics and Subtropics

Te Pas and Rees (2014) reported differences in yield between conventional and OA systems in the tropics and subtropics which were based on an extensive literature review. However, a metaanalysis was not undertaken, since this would mean that majority of the data would have to be left out due to missing information on standard errors, and it would have led to overrepresentation of a few studies with many observations, most of them located in a single region (mainly India). On average, yields were 26% higher under OA than those under conventional practices. Further, the highest yield increases in OA cropping systems were achieved in the least developed countries (116%), in arid regions (64%), and on coarse soils (142%). Among the 20 most common crop types, tomato, spinach (*Spinacia oleracea* L.), pepper, and lettuce produced much lower yields under OA. Otherwise, crops that are mainly produced in developing countries [eg, beans, millet, peanut (*Arachis hypogaea* L.), sorghum (*Sorghum bicolor* L. Moench) and maize], responded well to OA management. However, the data should be interpreted carefully. For example, organic amendments (manure, compost) may not be available on farms in similar amount to the ones applied in the reviewed controlled experiments. Further, observations were regionally clustered as ~40% of all data pairs originated from India but comparisons in Sub-Saharan Africa were underrepresented. In conclusion, OA farming in tropical and subtropical regions may be most successful in the least developed countries, in the driest regions, on coarse soils, and in systems that previously had low input levels (Te Pas and Rees, 2014).

5.2 Corn

At an experimental station in North Carolina, USA, a conventional–OA systems comparison was established in 1994 (Larsen et al., 2014). The conventionally managed plots outperformed organically managed plots by at least double during 2011 and 2012 in terms of sweet corn (*Z. mays* var. *saccharata*) yield. Specifically, in 2011 marketable yield was the highest for conventional NT plots (15.9 Mg ha⁻¹), followed by conventional CT plots (9.9 Mg ha⁻¹) with the conventional treatments yielding higher than the organic treatments (2.02 and 2.05 Mg ha⁻¹ for OA tillage and organic NT, respectively). Further, percentage of total yield that was marketable was much higher in the conventional treatments (75 and 86% for NT and tillage, respectively) compared to the organic treatments (39 and 42%). Weed competition was likely the major driver for the decreased yields in

the OA systems as, for example, OA-NT treatment had approximately 35 times more weed biomass C than the CT treatment. However, N limitation in OA systems may have also been a yield-limiting factor, based on observed yellowing of corn leaves in both study years. Further, due to poor legume growth, the majority of the cover crop stands in all treatment plots were dominated by wheat in 2011, providing low N contributions overall. The study demonstrated that conventional NT management results in an optimal combination of adequate sweet corn yields and high soil C status in humid, warm climates when a fall/winter cover crop is implemented (Larsen et al., 2014).

Sweet corn plant and ear fresh weight were lower for OA than for conventional systems initially after application of chicken manure compost at a long-term field trial in central Taiwan (8.2 vs 9.0 Mg ha⁻¹; Wang, 2014). However, no differences were observed between OA and conventional plots in the following 5 years. As total N absorbed by sweet corn plants and fresh ear weight correlated linearly, less N may have been available in chicken manure compost initially after application whereas at later stages N requirements of sweet corn plants were met (Wang, 2014).

Murrell and Cullen (2014) reported in a greenhouse study that corn yields did not differ between OA managed soil fertilized with dairy manure and 2 years of alfalfa, a similar OA treatment plus addition of gypsum (CaSO₄·2H₂O), and a conventionally managed soil. Also, yield losses caused by infestation with the European corn borer [*Ostrinia nubilalis* (Hübner)] did not differ among soil fertility treatments. The intraspecific competition between corn borer larvae had a stronger effect on insect performance than the fertilization regimes (Murrell and Cullen, 2014).

5.3 Rice

Meng et al. (2014) studied the effects of a conventional combination of mineral fertilizers with animal manure versus those of OA fertilizers [composted animal manure with castor (*Ricinus communis* L.) bean meal] on rice production at a site in Central Asia. By applying 802 kg N ha⁻¹, 80% of the conventional rice yield could be achieved by OA (7.6 Mg ha⁻¹ vs 9.4 Mg ha⁻¹). However, the conventional treatment received only 297 kg N ha⁻¹. It was concluded that a high-load application of OA manure was required to maintain high rice yields but the conventional treatment had a higher N utilization efficiency (29.6 vs 8.7–17.6% for different OA production systems, respectively; Meng et al., 2014).

OA and conventional farming of rice and giant river prawns (*Macrobrachium rosenbergii*) in rotational crops was tested in waterlogged paddy fields of Kuttanad, Kerala, India (Nair et al., 2014). Farming rice by OA reduced yields by 23%, from about 5.7 Mg ha⁻¹ in conventional farming compared to about 4.4 Mg ha⁻¹ in OA farming but those differences were not significant. The OA rice yield were similar to the many published reports and indicating a reduction in output (Nair et al., 2014).

A field experiment was conducted in Hyderabad, India, to compare conventional and OA rice production systems during 5 wet and 5 dry years (Surekha and Satishkumar, 2014). The organic amended sources were green manure, dhaincha (*Sesbania aculeata*), and paddy straw during wet seasons, and poultry manure and paddy straw during dry seasons. In the wet season, grain yields for the conventionally managed plots (5.3–5.5 Mg ha⁻¹) were 15–20% higher than those under OA during the first 2 years, and improved with OA (4.8–5.4 Mg ha⁻¹) in the later years to levels comparable to those of the conventionally managed plots. However, during the dry seasons yields at the conventionally managed plots (3.7–3.8 Mg ha⁻¹) were higher than those under OA (3.1–3.5 Mg ha⁻¹) for 4 consecutive years, and in the fifth year yields for OA (4.0 Mg ha⁻¹) were similar to those achieved at the conventionally managed plots (4.2 Mg ha⁻¹). This trend was partially explained by mismatch of nutrients released from organic amendments and crop demand as influenced by seasonal conditions in the initial years. Once the soil fertility was built up sufficiently, yields under OA and conventional practice were similar. Thus, repeated application of organic amendments over the years have the potential to build up sufficient soil fertility by improving soil biological activity (Surekha and Satishkumar, 2014).

At a long-term field trial in central Taiwan, rice grain yields did not differ in 4 out of 5 years between conventional (5.65–6.26 Mg ha⁻¹) and OA practices (5.37–6.15 Mg ha⁻¹; Wang, 2014). However, in the first study year yields were lower in the OA compared to the conventional system (5.37 vs 6.62 Mg ha⁻¹, respectively). Apparently, only a fraction of the chicken manure compost N was initially available to rice but overall the N requirement of rice was met by the amount of N in the compost (Wang, 2014).

5.4 Wheat

Mean dry matter winter wheat yields were lower for an OA treatment (N source symbiotically fixed N) in Estonia compared to a conventional treatment receiving mineral N fertilizer in the first 4 years after establishing

the experiment (2.3 vs 4.3 Mg ha⁻¹; Alaru et al., 2014). However, when cover crops, or cover crops and manure were added to the OA system, yields were not significantly different from those under the conventional system (2.9 Mg ha⁻¹). Alaru et al. (2014) recommended to apply organic amendments with faster mineralization rates and splitting the application of organic N in the crop cycle period to supply sufficient quantities of N during rapid plant growth for increasing winter wheat yields under OA. Similarly, limited supply of available N in organic fertility management systems in northeast England contributed to lower wheat yields at OA compared to conventional systems during 4 years (on average 4.8 vs 7.9 Mg ha⁻¹; Bilsborrow et al., 2013). To improve yields in OA production systems, Bilsborrow et al. (2013) suggested to focus on improving the fertilizer use efficiency from organic amendments and fertility building crops via breeding (selection of varieties with higher N uptake efficiency from organic amendment inputs) and agronomic approaches (eg, use of split dose application of organic fertilizers and/or the use of organic amendments with a higher content of readily available forms of N).

N supply and weed control were identified as the main factors constraining durum wheat (*T. durum* Desf.) grain yield for OA systems under Mediterranean climate conditions, especially when excess of rainfall and low temperatures occurred throughout the crop reproductive period (Campiglia et al., 2015). Specifically, averaged over 6 years the durum grain yield was 15% lower in OA compared to conventional systems (2.86 vs 3.40 Mg ha⁻¹, calculated from data in Campiglia et al., 2015), 4 years after plot establishment. However, the yield gap between the OA and conventional cropping systems varied from -5% to -32% across the years, and changes in weather conditions may have contributed to this variability (Campiglia et al., 2015).

Mean winter wheat grain yields at 34 trials for the period 2004–11 in north-west France were 7.26 Mg ha⁻¹ for conventional low-input systems and 5.19 Mg ha⁻¹ for OA (Le Campion et al., 2014). The yield gap at the different locations ranged between 25% and 40%, and N supply appeared to be the main factor explaining the differences among locations (Le Campion et al., 2014).

OA systems cropped to winter wheat attained on average 64% of the conventional yields between 25 and 32 years after establishment of a field experiment in Switzerland (3.5–3.8 Mg ha⁻¹ vs 5.6–5.8 Mg ha⁻¹, respectively; Mayer et al., 2015). The effects of the preceding crop potatoes (*Solanum tuberosum* L.) in comparison with preceding silage maize

outperformed the organic amendment effects, resulting in 33% higher yields for the OA systems. Thus, a more synchronized nutrient supply throughout the wheat development due to the preceding potatoes could reduce the yield gap between OA and conventional systems (Mayer et al., 2015).

N shortage may have been the reason why OA wheat yielded less than conventional wheat, 9 years after the establishment of field experiments at a Mediterranean site in Italy (Mazzoncini et al., 2015). Specifically, grain yields and total dry matter yields were 2.65 and 5.82 Mg ha⁻¹ for organic and 4.85 and 10.17 Mg ha⁻¹ for conventional systems, respectively.



6. IMPLICATIONS OF ORGANIC AGRICULTURE FOR THE ENVIRONMENT

The OA practice has implications for the environment at the field, farm, regional, and global scales (Stockdale et al., 2001). It is generally thought that OA is environmentally benign compared to conventional agriculture because insecticides, herbicides, and chemical fertilizers are entirely or largely avoided. However, soil, air, and water quality, and biodiversity may be particularly affected by any agricultural practice, and those effects together with energy use and land requirements will be discussed in the following sections.

6.1 Soil Quality

Soil quality (ie, soil's ability to deliver ecosystem services) may be affected by OA. Studies summarized by Gomiero et al. (2011) and Gomiero (2013) have shown that OA performs better in preserving or improving soil quality with regards to both biophysical (ie, stored nutrients) and biological (ie, biodiversity) properties. For example, higher soil quality in OA systems, particularly enhanced C and N storage, was reported for six long-term OA comparison sites in the United States (Delate et al., 2015). However, available K and P levels may also be lower in soils under OA than those under conventional management practices (Stockdale et al., 2001). Otherwise, OA managed soils may have a much higher water-holding capacity than conventionally managed soils (Gomiero et al., 2011). Especially under drought conditions, both improved water capture and water-holding capacity of soil under OA management can contribute to higher crop yields. Soil loss by erosion may also be greatly reduced under OA management (Gomiero et al., 2011). Additions of

manure and other OA residues increase soil aggregation (Lynch, 2014). Thus, aggregate stability may be higher for OA than conventionally managed soils, and, thus, soils under OA may be less susceptible to erosion. However, more stable organic amendments such as compost may also have less effects on aggregation and the susceptibility of soil to erosion compared to readily available organic amendments (Lynch, 2014).

Soil biochemical and ecological characteristics appear to be improved by OA (Gomiero et al., 2011). Specifically, OA farming performs much better than conventional farming with regard to root length colonized by mycorrhizae. Further, OA farming performs better or much better than conventional farming with regard to microbial biomass and activity, and soil biodiversity (Gomiero et al., 2011). For example, several studies have indicated that OA is associated with higher levels of biological activity, represented by bacteria, fungi, springtails, mites, and earthworms (Gomiero, 2013).

The soil microbial community composition appears responsive to effects of farming system per se, and key constituents of the soil microbial community may be sensitive to field crop management regimes (Lynch, 2014). However, it is important to note that soil characteristics are generally site-specific and local specificity plays an important role in determining the performance of any farming system (Gomiero, 2013). Further, differences between management practices within a given farming system can be as or more influential than the effects of the farming system per se (Lynch, 2014). Thus, nonsystematic differences in soil properties between conventionally and OA managed soils have also been reported. For example, some OA systems may perform similar to conventional systems regarding soil biophysical characteristics, but worse regarding soil biology (Gomiero et al., 2011). With regard to effects on pest control, OA systems performed the same, better, or much better than conventional systems. Further, OA did not systematically influence many physical and chemical properties at two conventional–OA pairs of grassland farms in Iceland and two pairs of arable farms in Austria (van Leeuwen et al., 2015). Specifically, the soil aggregate size distribution was consistently higher on OA than on conventional farms in Iceland, but no differences were found in Austria. Further, C and N mineralization rates, stocks of hot-water-extractable C, total N and potential mineralizable N, and bacterial activity were quite similar for both farming systems. Also, soil food web structures, in terms of presence of trophic groups of soil organisms, were highly similar among all farms. However, soil organism biomass, especially of bacteria and nematodes, was consistently higher on

OA than on conventional farms. Within the microarthropods, taxonomic diversity was systematically higher in the OA farms compared to the conventional farms. Thus, OA can enhance soil organism biomass whereas chemical and physical soil properties may not consistently differ between organic and conventional systems (van Leeuwen et al., 2015).

Soil health is defined as “the capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health” (Doran and Zeiss, 2000). Key indicators of changes in soil health are soil organisms, including the abundance and diversity of bacteria, fungi, and nematodes, as they respond sensitively to anthropogenic disturbance (Lynch, 2015). However, the soil microbial community diversity shows a high degree of resilience to farming system management compared to the influence of temporal shifts or specific crop sequence influences. Thus, soil health benefits by OA appear consistently achieved only for larger soil organisms. Otherwise, functional properties such as enhanced biochemical and biological turnover of organic P appear to be enhanced by OA farming systems, and may contribute to enhanced P use efficiency (Lynch, 2015).

In summary, some studies reported improvement in soil physical properties under OA practices while others found no differences between conventional and OA systems (Stockdale et al., 2001). The knowledge about the effects of OA farming systems on soil ecology has been greatly improved but the link between changes in soil ecology under OA practices and specific agroecosystem services needs additional research (Lynch, 2014). The core premises of consistent benefits to soil health by OA via enhanced microbial diversity has been challenged (Lynch, 2015). Otherwise, specific cropping practices and production system intensity overall, rather than farming system per se, may influence both nutrient cycling and soil ecosystem functioning, and effects of practices and intensity need additional research.

6.2 Air Quality

Similar to conventional agriculture, OA may contribute to air contamination by releasing nutrients, heavy metals (ie, Cu), pathogens, particulate matters, and noxious gases into the atmosphere (Udeigwe et al., 2015). For example, the widespread OA farming practice of CT may impair air quality by releasing fine dust and debris, as this practice involves multiple tillage passes and more surface soil disturbance compared to conservation tillage.

Otherwise, the generally reduced soil erosion on organically managed farms may result in lower air-borne particulate concentrations compared to those of conventionally managed farms (Gomiero, 2013). Further, air pollutants such as particulate matters (PM_{2.5}, PM₁₀), oxides of N, C, and S, as well as NH₃, CH₄, H₂S, volatile organic compounds, and pathogens have been commonly tied to processing and surface application of animal manures, and emissions from feedlots (Udeigwe et al., 2015). The air quality may also be worsened by OA due to N losses from organic compost or green manures through volatilization of surplus N failing to match crop demand (Gomiero, 2013).

6.3 Water Quality

The scientific literature on water quality in OA systems is rather limited (Cambardella et al., 2015). OA may perform much better than conventional agriculture with regard to surface and ground water quality by halting the use of harmful chemicals, that is, conventional synthetic pesticides (Gomiero et al., 2011). Otherwise, water quality risks by pesticides permitted in OA are largely unstudied (Stockdale et al., 2001). With regard to nitrate leaching, OA systems may perform differently as, on one hand, N uptake efficiency may be enhanced at some organically managed sites (Gomiero et al., 2011). On the other hand, N losses through leaching from organic amendments may be increased at other sites as N release from organic compost or green manures may fail to match crop N demand (Gomiero et al., 2011). A key to reducing nitrate leaching from OA systems is the management of residual N from legumes (Stockdale et al., 2001). Appropriate timing of tillage is the most influential tool available to farmers to manage N synchrony. Tillage that incorporates organic N inputs preceding cash crops can promote synchrony of N mineralization and crop demand, while late or postseason tillage will promote nitrate leaching by stimulating soil inorganic N pulses that are asynchronous with plant uptake (Finney et al., 2015). In subsurface-drained landscapes, OA farming practices such as the application of composted animal manure, and the use of forage legumes and green manures within extended cropping rotations, can improve water quality (Cambardella et al., 2015). For example, averaged over a 3-year period subsurface drainage water nitrate-N loss from conventionally managed corn-soybean systems in the Midwestern US was nearly twice as much (79.2 kg N ha⁻¹) as that from organically managed corn-soybean-oats/alfalfa (39.9 kg N ha⁻¹; Cambardella et al., 2015).

6.4 Biodiversity

The benefits of OA farming for biodiversity in agricultural landscapes are under discussion. A hierarchical metaanalysis based on [Bengtsson et al. \(2005\)](#) and updated by including 194 observations from 94 studies indicated that OA farming increased species richness by about 30% ([Tuck et al., 2014](#)). However, the effects varied with the organism group and crop studied, and with the proportion of arable land in the surrounding landscape. For example, at the local scale the abundance and species richness of tachinid parasitoids was higher for OA compared to conventional farms ([Inclán et al., 2015](#)). At the landscape scale, the diversity of tachinids was higher in landscapes with higher proportions of land under OA management. Further, the positive effect of OA farming on tachinid parasitoids diversity was clear on both scales for arable fields but not for grasslands ([Inclán et al., 2015](#)). Larger effects of OA were generally observed in cereals, among plants and pollinators, and in landscapes with higher land-use intensity ([Tuck et al., 2014](#)). Plants benefited most from OA probably because of restricted herbicide use. However, despite the fact that OA farming has been shown to have large effects on soil properties, its effects on decomposers and soil organisms were ambiguous ([Tuck et al., 2014](#)). For example, increasing rotational diversity by OA may fundamentally change soil microbial community structure and activity, with positive effects on aggregate formation and SOC accrual ([Tiemann et al., 2015](#)). Otherwise, OA increased bacteria and not fungi in a long-term experiment in France ([Henneron et al., 2015](#)). Mainly the bacterial pathway of the soil food web, and endogeic and anecic earthworms were improved under OA. Further, macrofauna, nematodes, and microorganism abundance and/or biomass were increased by OA but not predaceous nematodes ([Henneron et al., 2015](#)). Otherwise, at an experimental site in the United States, soil microbial traits did not differ between OA and conventional management practices ([Wickings et al., 2016](#)).

The soil microbial community composition may be resilient to changes in farming system, and variation in soil type and structure may be more important for soil organisms in general than the farming system itself ([Lynch, 2015](#)). Most importantly, whether OA can reduce environmental impacts, enhance crop yield and result in a more sustainable agricultural system by favoring a rich and abundant soil life needs to be confirmed by field studies ([Bender and van der Heijden, 2015](#)). To date, studying the effects of OA farming on biodiversity has been heavily biased toward agricultural systems in the developed world, especially Europe and North America. Thus, more

studies are particularly needed in tropical, subtropical, and Mediterranean climates. Nevertheless, given the large areas of land under agricultural production, OA farming methods may play a major role in halting the continued loss of biodiversity from industrialized nations. However, as yields are also lower under OA (Ponisio et al., 2015), more wild or marginal land may have to be brought into agricultural production to produce the same amount of biomass and food. This land is likely to have supported even higher biodiversity than OA farms. Thus, it is unclear whether there is an overall benefit or cost of OA farming to biodiversity (Tuck et al., 2014).

In addition to terrestrial biodiversity, OA may also affect the biodiversity of aquatic ecosystems. For example, the inorganic fungicides Cu and elemental S authorized in OA farming caused both structural and functional changes in leaf-associated microbial communities in surface waters (Zubrod et al., 2015). Any effect on microorganisms involved in leaf litter breakdown in aquatic ecosystems may have far-reaching consequences for the detritus-based food web due to its bottom-up regulation. At concentrations measured in surface waters adjacent to OA farming fields, Cu may affect aquatic bacteria and fungi as well as their functions (Zubrod et al., 2015).

6.5 Energy Use

Agriculture and food systems play an important role in fossil fuel consumption and climate change because of their significant energy use, for example, fossil fuel use for operating machinery, for the management of agricultural soils, and the production of synthetic fertilizers (Smith et al., 2014). OA farming is favorable with respect to whole-farm energy use and energy efficiency both on per hectare and per farm product basis, with the possible exception of poultry and fruit sectors (Lynch et al., 2011). However, based on calculations of net energy production per unit area, Bertilsson et al. (2008) showed that conventional systems produce far more energy per hectare, that is, more solar energy is bound compared to OA systems. The highly increased crop production when using N fertilizer results in a very positive energy balance, with at least a sixfold return on the energy invested for N fertilizer production. Otherwise, growth of legumes for biological N fixation instead of chemical fertilizer production does not improve the energy budget of OA systems. Lower yields in the OA systems, and consequently lower energy production per unit area, mean that more land would be required to produce the same amount of energy. This greater land requirement in organic production must also be considered in calculating energy

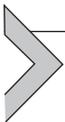
balances of agricultural systems (Bertilsson et al., 2008). In addition, for comparing conventional and OA systems, not only agriculture production but also postharvest practices and distribution networks and the energy consumption therein must be considered (Ziesemer, 2007). However, little information is available regarding differences in energy use between OA and conventional processing, packaging, storage, and distribution. There is some evidence that OA systems may offer overall less energy intensive methods than their conventional counterparts. Specifically, OA systems utilize less energy than conventional agriculture production due to less reliance on energy intensive fertilizers, chemicals, and concentrated feed (Ziesemer, 2007).

Based on a review of 50 studies, Smith et al. (2015) suggested that OA farming performs better than conventional for nearly all crop types when energy use is expressed on a unit of area basis but results are more variable per unit of product due to lower yields for most organic crops. Ruminant production systems tend to be more energy efficient under OA management due to the production of forage in grass–clover leys. Conversely, organic poultry tend to perform worse in terms of energy use as a result of higher feed conversion ratios and mortality rates compared to conventional fully housed or free-range systems. OA farming also performs worse for potatoes, where a lower yield reduces efficiency, and other vegetables that require flame weeding. Further, there is some evidence that OA farms use more renewable energy. In addition, human energy requirements on OA farms are higher as a result of greater system diversity and manual weed control (Smith et al., 2015). Otherwise, OA systems use less of the energy-demanding implements such as irrigation, heavy machinery, and heated greenhouses (Ziesemer, 2007). Overall, the energy efficiency of most cropping and ruminant livestock farming systems may be enhanced through the adoption of OA management practices. However, in many cases this will be at the expense of crop or livestock yields (Smith et al., 2015).

6.6 Land Requirement

Several reviews and metaanalysis have shown that OA yields are lower than those achieved under conventional agricultural practices. Thus, OA relies on more land to produce the same amount of food compared to conventional agriculture. For example, relative to conventional systems, between 9% and 214% more land is needed to produce one unit arable crop by OA, and

between 6% and 346% more land is needed to produce one unit animal product (Meier et al., 2015). In addition, adopting OA on a large scale could potentially also threaten the world's forests, wetlands, and grasslands (Crowder and Reganold, 2015). Further limitations to land conversion for OA may also arise from the projected urban expansion. While more than 43-Mha land is currently under OA, urban areas may expand by 120 Mha by 2030 (Seto et al., 2012). Some argue that the inefficiency of OA will make it less relevant in the future as global food security must be achieved in a world suffering from climate change and rapid population increase (Pickett, 2013). Nevertheless, the demand for organic food is growing and with it the land area under OA although at a small scale relative to that of conventional agricultural land (Willer and Lernoud, 2015).



7. CONCLUSIONS

All agricultural systems inevitably impact the environment but OA systems are perceived as having less deleterious effects than conventional systems. However, scientific evidence for those environmental advantages is inconclusive. OA is less than a century old with the first distinct form of OA introduced by the Austrian philosopher Rudolf Steiner. Since then, the global area under OA has grown driven by consumer demand, especially, in Europe and the United States but only 1% of global agricultural land is currently farmed by OA methods. However, OA may cause a reduction in soil profile SOC stocks as plant-derived C inputs are lower because of reduced yields and as tillage is often applied for weed control, but effects on SIC stocks are less well known. Further, there is some evidence that soils under OA emit less CO₂, N₂O, and CH₄ than conventionally managed soils but direct measurements are scanty and geographically biased toward studies from Europe. In comparison to conventional, organic yields are on average about 19% lower as soluble mineral fertilizer inputs are prohibited, and synthetic herbicides and pesticides are rejected by OA. While soil, air, and water quality may be enhanced under OA and energy use lower compared to conventional practices, effects of OA on biodiversity are debatable. In the future, an increasing share of agricultural land will be farmed by organic methods as consumer demand continues to grow but long-term field experiments in major global agricultural regions accomplished by LCA are needed to more comprehensively assess the environmental impacts of OA.

7.1 Pros

- Lower emissions of CO₂, N₂O, and CH₄
- Enhanced soil and water quality
- Lower energy use per land area
- Higher energy efficiency per land area

7.2 Cons

- Lower soil profile SOC stocks
- Lower crop yields
- Higher land requirement
- Lower energy production per land area

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