THE INFLUENCE OF MANAGEMENT PRACTICES ON THE GREENHOUSE GAS BALANCE OF MEDITERRANEAN CROPPING SYSTEMS. IDENTIFYING THE CLIMATE CHANGE MITIGATION POTENTIAL THROUGH QUANTITATIVE REVIEW AND LIFE CYCLE ASSESSMENT

DOCTORAL THESIS

EDUARDO MANUEL AGUILERA FERNÁNDEZ

UNIVERSIDAD PABLO DE OLAVIDE

DEPARTAMENTO DE GEOGRAFÍA, HISTORIA Y FILOSOFÍA

LABORATORIO DE HISTORIA DE LOS AGROECOSISTEMAS

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Doctoral thesis

Eduardo Manuel Aguilera Fernández

Supervisors

Gloria Isabel Guzmán Casado

Luis Lassaletta Coto

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We hereby declare that the Doctoral Thesis entitled "The influence of management practices on the greenhouse gas balance of Mediterranean cropping systems. Identifying the climate change mitigation potential through quantitative review and life cycle assessment" has been carried out by Eduardo Manuel Aguilera Fernández and directed by ourselves.

This Thesis has our approval and it is ready for its oral presentation and defense.

September 20, 2016.

Luis Lassaletta

Gloria Isabel Guzmán Casado

Luis Lassaletta Coto

A mis padres

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Abstract

Resource depletion and global change trends require urgent action in order to mitigate the change and adapt to them. In particular, agricultural GHG emissions are significant contributors to climate change. Mediterranean cropping systems are highly vulnerable to climate change and, at the same time, they are an important source of GHG emissions. The scientific information on GHG emissions in Mediterranean systems is growing, but there is need to systematize and integrate the knowledge.

In this PhD dissertation, the two main soil processes responsible for the GHG balance, N_2O emissions and C sequestration, are studied through qualitative review and meta-analysis of published information under Mediterranean climate conditions (Studies 1 and 2). In studies 3 and 4, a life cycle assessment (LCA) of 80 organic and 80 conventional farms was performed, including all processes involved in the GHG emission balance, and employing climate-specific coefficients derived from the previous meta-analyses for the calculation of N_2O emissions and C sequestration.

The results show distinct GHG emission patterns in Mediterranean cropping systems. Study 1 shows that N_2O emissions from rainfed systems are much lower than the global IPCC value, while drip irrigation systems seem a promising N_2O mitigation strategy under irrigation. Solid organic fertilizers are related to lower N_2O emissions than synthetic fertilizers. Study 2 shows that carbon sequestration is highly responsive to management changes. Best performing practices are those associated to the highest C input application rates, including organic farming practices. It is possible to achieve relatively high C sequestration rates using internal C inputs such as crop residues and cover crops. Studies 3 and 4 showed that the organic farming systems were generally associated to lower total GHG emissions both on a surface and on a yield-scaled basis, mainly due to the non-use of synthetic fertilizers and to carbon sequestration enhanced by the application of organic fertilizers. In some cases, carbon sequestration under organic farming was enough to offset all other GHG emissions, leading to carbon-neutral cropping systems, while in others, lower yield affected the performance of organic systems. The GHG balance of most studied systems is dominated by energy use (including indirect energy from fertilizer manufacture in conventional systems) and C sequestration, which indicates that these are the two processes in which most mitigation efforts should be focused.

Chapter 1. Introduction

1.1. Agriculture in a changing world

Since its appearance in various regions of the world around 10,000 years ago (Barker, 2006), agriculture increased the energy humans obtain from ecosystems, allowing population and complexity growth (Smil, 2013, González de Molina and Toledo, 2014, Fernández Durán and González Reyes, 2014). It also became the main way in which humans interact with the environment. During the 20th century, agriculture experienced drastic transformations, transitioning from a solar-based, "organic" metabolism to another based on fossil fuels and new technologies. The huge growth in agricultural production achieved was accompanied by a dramatic growth in environmental impacts, which were added to other impacts of industrialization and population expansion. The magnitude of these human-driven changes in the Earth systems is so large that they are giving place to a new geologic era, the Anthropocene (Steffen et al., 2011). On the other side of the coin, the coming decades are expected to be shaped by the impact of reaching the limits to resource extraction. As a result of both processes, social-ecological resilience is challenged, calling for a fundamental shift in world views and institutions (Folke et al., 2011). Thus, in the 21st century we have entered an entirely new situation for agricultural systems, which now rely on diminishing resources, are forced to adapt to global changes and need to reduce their environmental impacts while still being able to feed a growing human population (Foley et al., 2011).

Despite the novelty of its global scale, the present situation resembles others already experienced by agriculture in mature stages of many other civilizations (Tainter, 1988): the need to feed and supply goods for a large and "sophisticated" population in a context of diminishing returns due to degradation of the funds that support it. The difference with previous historical moments is a matter of scale: the population of the present industrial civilization is now settled along the entire world and its size is orders of magnitude higher than the largest of ancient civilizations. The growth rates are impressive: while global population barely reached 1 billion in 1800 (UN, 1999), in 2011 it surpassed 7 billion inhabitants (UN, 2015) in a tightly interconnected world, and projections suggest it could reach 10-12 billion by 2100 (Gerland et al., 2014). On the other hand, the level of "sophistication" of this population (i.e. its per-capita metabolic profile), with average people in industrial societies consuming 3-5 times more energy and materials than in agrarian societies (Fischer-Kowalski and Haberl, 1997, Krausmann et al., 2008), while being able to travel thousands of kilometers regularly or to process billions of bytes in seconds, could not have been even imagined a few generations ago. In terms of demand of agricultural products, two current expressions of this sophistication are the increasing content of animal products in global diets (Kastner et al., 2012, Billen et al., 2014, 2015, Tilman and Clark, 2014) and the production of biofuels (Lotze-Campen et al 2010). The increasing dominance of ultra-processed food is another worrisome trend in the global agri-food system (Monteiro et al., 2013), with negative consequences for health and energy consumption. Likewise, modern agri-food systems are characterized by very high waste rates, particularly at the consumption level (Gustavsson et al., 2011), which is associated to environmental problems such as nitrogen pollution (Grizzetti et al., 2013).

The combination of continued growth in population and per-capita consumption has resulted in an increase in global materials use by 8-fold in the 20th century (Krausmann et al., 2009). In parallel, the environmental degradation process faced by the current civilization has reached, as well as its own metabolism, a global scale. In the present, not even the most remote marine ecosystems are absent from human alteration (Halpern et al., 2008). For the first time in History, humanity is threatening the ecological mechanisms maintaining the homeostasis of the planet, which is expressed as systemic disruptions of major planetary systems such as the climate, biodiversity or nutrient cycles (Röckstrom et al., 2009, Steffen et al., 2015). Said in another way, the capacity of the Earth to assimilate the residues of the huge, linear metabolism of modern civilization is reaching its limits. The reverse of this problem caused by an oversized, dysfunctional metabolism is the depletion of the resources of which it is fed: energy and materials.

Thus, global agriculture, as the major source of food for humanity, has reached a crossroad in the 21st century, a trilemma that could be defined by its three major challenges: production, adaptation and mitigation.

i) produce enough quality food for a growing population, while
ii) adapting to global change, this is, having less resource availability (e.g. energy, phosphorus or water) and altered planetary systems, particularly climate change
iii) with less socio-environmental impacts, including lowering greenhouse gas (GHG) and pollutants emissions and promoting human health

1.1.1 Reaching planetary limits and the role of agriculture

Depletion of natural resources: Peak everything

The concept of **peak oil** was coined by M. King Hubbert (1956) in his projections of US oil production. The basic idea is that the production of an oil well, field or region follows a bell-shaped curve, in which the peak is reached when approximately half of the resources have been extracted. This pattern could also be applied to the world as a whole. The acceptance of this idea within the scientific community and the general public is difficult, as it has profound and unpleasant social and economic implications (Bardi, 2009). Despite there are opposing views (e.g.

Radtezki, 2010, Towler, 2011), most studies agree that global peak oil could be reached in the next few years, or could even have already been reached (e.g. Nashawi et al., 2010, Sorrell et al., 2010, Gallager, 2011). A clear symptom of the proximity of peak oil is the "phase shift" in oil economics since 2005, with production not rising significantly despite major increases in oil prices (Murray and King, 2012). Low oil prices in the 2014-2016 period could be seen as a rebuttal to oil limits to economic growth. Nonetheless, Tverberg (2012) already warned that oil limits may also appear as an "oil glut": an excess of high-priced oil in the market leading to a constraint in demand due to the weakening of the global economy. In a context of high production costs, the resulting low oil prices are causing an impressive cut back in oil exploration and development investment (AlixPartners, 2016), which is expected to lead to steep reductions in oil supply in the coming years (HBSC, 2016).

A major implication of the depletion of oil and other resources is the decline in their Energy Return On Energy Invested (EROI), meaning that an increasing amount of energy has to be diverted to energy production to sustain consumption levels (Murphy and Hall, 2010). This process can also have an impact on production rate, indicating that capital investment has not only to be maintained, but it has to increase in order to avoid steep production declines (IEA, 2014). As will be discussed below, oil depletion and declining EROIs can exacerbate environmental problems associated to oil extraction.

Oil is not the only fossil fuel being depleted: **peak gas** is expected for 2025-2066 (Mohr and Evans, 2011), while **peak coal** might have already occurred in the US (Milici et al., 2013) and is expected as early as 2024 in China, the world major coal consumer (Wang et al., 2013), with world peak coal estimations ranging from 2011 (Patzek et al., 2010) to beyond 2100 (Thielemann and Schiffer, 2012).

Furthermore, not only fossil energy resources are being depleted by human activities: the "peaks" of many **metallic and non-metallic minerals**, such as iron, aluminum, copper, phosphorus, etc. are expected by some authors before the end of the 21st century (Valero and Valero, 2010). But the story of resource depletion does not end within non-renewable resources. Even **renewable resources** are approaching their peaks, have already peaked or have even collapsed in many areas due to extraction rates larger than natural recovery rates, including water (Gleick and Palaniappan, 2010), fisheries (Cardinale et al., 2012a, Anderson et al., 2011) or timber (Shearman et al., 2012).

Some of the resources being depleted are **agricultural inputs**: for example, crop fertilization is the main use of **phosphorus** (P). Estimations of P peak date vary widely between 2030 (Cordell, 2009) and beyond 2100 (Van Vuuren et al., 2010). In any case, future P scarcity and increasing production costs will interact with other factors related to global change and resource depletion, increasing the P vulnerability of food production systems (Cordell and Neset, 2014, Obersteiner et al., 2013), particularly due to an increasing reliance on just a single country possessing the majority of world phosphate reserves, Morocco (Walan et al., 2014). Diminishing **water** resources is also an important concern for food security, taking into account that irrigation is responsible for >10% of global cropland NPP (Ozdogan, 2011), and particularly in a context of climate change (Hejazi et al., 2014). At the global level, the groundwater footprint is about 3.5 times the actual area of aquifers, indicating a high degree of over-exploitation and a serious threat to ecosystems and human populations that depend on them (Gleeson et al., 2012). Indeed, irrigation in many areas is increasingly based on fossil water resources, which is associated to vast increases in energy consumption (Karimi et al., 2012). Decreasing water levels due to continued exploitation of these aquifers implies further increases the energy needs of water extraction (Li et al., 2013).

Resource depletion will severely impact the **economy**, undermining economic growth and potentially leading to zero or negative growth rates (Murphy and Hall, 2011, Capellán-Pérez et al., 2014). The high initial investment of renewable energy, as a result of a long payback time, imposes an important energetic and emissions burden (Kawajiri et al., 2014). The impact of the end of cheap oil caused the stagnation of labor productivity in the US, leading to a heavy reliance on debt increase for the continuation of economic growth (Kaufmann, 2014), while at the same time, decreasing oil availability and increasing oil extraction costs are making more and more difficult to repay the debt (Tverberg, 2012). The increasing trend and volatility of oil prices will also negatively impact international trade, potentially leading to a reversal of globalization (Curtis, 2009, Chen and Hsu, 2012). This way, the world economy seems to follow the collapse path predicted by the 1972 Club of Rome report (Meadows et al., 1972, Turner, 2012). Thus, fossil fuel-dependent global agriculture (Woods et al., 2010) would have adapt not only to shortages of agricultural inputs caused by resource depletion, but also to the profound impacts that this decline will have on the economy.

Alteration of planetary systems: global change

Global change is characterized by the simultaneous global alteration of various relevant components of the planetary system (Vitousek, 1994). Thus, climate change is just one of the many planetary systems threatened by human activity. Rockström et al. (2009) outline three other systems that could be trespassing the boundaries of the safe operating space for humanity. Crossing those boundaries implies that unpredictable cascading effects on Earth system properties are likely to occur, potentially leading to planetary-scale regime shifts with associated abrupt and massive changes (Scheffer and Carpenter, 2003; Hughes et al., 2013). According to Rockström et al. (2009), the biodiversity loss and the alteration of the nitrogen cycle are already beyond that threshold, while other important homeostatic systems such as the phosphorus (P) cycle, water use, land use and ocean acidification are close to the safety limit. Steffen et al. (2015)

also include the alteration of the phosphorus cycle beyond their planetary boundary, and climate change and land-system change already in the zone of uncertainty, whereas the boundary for the nitrogen cycle is re-defined. These alterations in global systems interact between them, usually reinforcing their negative effects in a non-linear way (Drijfhout et al 2015). I will discuss the nitrogen cycle in Section 1.2.2, global warming in the next-sub-section, and the effects of interactions in the following sub-sections. In the following paragraphs I overview some other components of global change.

Regarding **biodiversity loss**, increasing evidence supports the idea that a sixth mass extinction could be under way (Banorsky et al., 2011), a trend that is not slowing down despite international efforts to reduce the rate of biodiversity loss (Butchart et al., 2010), which is now about 1,000 times the likely background rate of extinction (Pimm et al., 2014). A worrisome consequence of biodiversity loss are the associated ecosystem changes (Hooper et al., 2012), such as the reduction in the efficiency by which ecological communities metabolize resources, including their capture from the environment, their transformation into biomass and their recycling (Cardinale et al., 2012b). Consequently, the productivity of ecological communities is also dependent on their biodiversity levels (Tilman et al., 2012). A related issue is the relationship between biodiversity and stability of ecosystem functions and the provision of ecosystem services. The available research evidence overwhelmingly suggests that biodiversity loss can severely impact basic ecosystem services, including provisioning (production of food, feed and materials) and regulating (biocontrol of pests and diseases, carbon cycle controls of climate, soil and water quality, or pollination) services (Cardinale et al., 2012b). Thus, agriculture is severely threatened by biodiversity loss. At the same time, land use change to produce world traded commodities such as food, feed, and timber is the major driver of biodiversity loss (Chaudhary et al., 2016). At the same time, however, diversified agroecosystems can contribute to biodiversity conservation (Tscharntke et al., 2012).

The human alteration of the **P cycle** is mainly driven by crop-livestock production systems. P budgets were almost balanced in 1900, but global P surplus raised to 2 Tg/yr in 1950 and 11 Tg/yr in 2000; while P withdrawals only tripled, P inputs increased 5-fold (Bouwman et al., 2013a). A large share of this increase was due to P fertilizer input, which was very small in 1900 and represented almost half of P input in 2000. The implication of this dependence on non-renewable P resources is discussed in the previous sub-section.

The **potassium (K)** cycle has usually be neglected in global change studies, but the importance of this element for plant growth, and the magnitude of the human alteration of its cycle, suggest that it should be considered in ecological models, particularly in arid and semi-arid ecosystems (Sardans and Penuelas, 2015).

Despite land use change has being traditionally considered a local issue, the magnitude of the changes have led it to become a force of global importance (Foley et al. 2005). **Land-system change** is quantified in the planetary boundary approach by the amount of cropland (Rockström et al., 2009) or the amount of forest remaining (Steffen et al., 2015). The boundary is higher for tropical and boreal forests (85% loss) than for temperate forests (50% loss) because tropical forests have a large influence on the climate system through evapotranspiration, and boreal forests through albedo, while the effect of temperate forests is weaker (Steffen et al., 2015). Global total forest loss over the period 2000-2012 exceeded 5% of forest cover (Heino et al., 2015), and was mainly driven by agricultural (cropland and grassland) expansion, which is taking place at a high rate in all major tropical forested regions such as South America (De Sy et al., 2015) and South-East Asia (Henders et al., 2015). In turn, this expansion is driven by increased demand of agricultural commodities, particularly meat and animal feedstuff (Henders et al., 2015). **Land degradation,** is a serious problem in many areas, caused among other factors by deforestation, unsuitable tillage, fertilization and irrigation management, and can potentially have drastic socioeconomic consequences such as migration (Kapur et al., 2006).

The **water cycle** is now deeply altered by human activity, which is now the dominant force driving changes in global water resources (Rockstrom et al., 2014). Indeed, the planetary boundary for freshwater use is being approached rapidly (Gerten et al., 2013). The human impact on the water cycle is not only due to the regulation of rivers with dams and channels in order to allow human consumption of blue water, but also to human alteration of rainfall stability (both due to land cover change and to global climate change), groundwater overexploitation and water pollution (Rockstrom et al., 2014). Agriculture affects **water security** and **freshwater biodiversity** through multiple drivers of stress, including some related to watershed disturbance (such as cropland or livestock density), other to pollution (salinization, N, P, pesticide, sediments and organic loadings through leaching or erosion), other to water resource development (dams, river fragmentation, consumptive water loss, human and agricultural water stress) and biotic factors (aquaculture pressure) (Vörömarty et al., 2010).

Ocean acidification is mainly caused by increased atmospheric carbon dioxide (CO_2) concentrations and dissolved inorganic carbon resulting from mineralization of vegetal biomass and dissolved organic matter, which are enhanced by nutrient exports from rivers and their associated effects. The negative impact of ocean acidification on marine organisms is amplified by many other phenomena generated by global change such as eutrophication, ocean warming or pathogen outbreaks (Mostofa et al., 2016).

Climate change: one component of global change

According to the last IPCC report (IPCC, 2013), which synthetizes the latest scientific evidence about the climate system, global warming is an unquestionable process, as shown by the increase in global temperatures of air and water, the retreat of ice and snow, and the rise in sea level. The changes at regional levels are affecting many natural ecosystems, mainly through the increase in temperature and the changes in precipitation patterns.

Average **global temperature** did not increase significantly in the 2000-2013 period. This has been called the global warming "hiatus" (e.g. Kaufmann et al., 2011), potentially challenging climate change theories (see Lewandowsky et al., 2015). However, this trend can be explained by multiple factors considered in current climate models, including natural forcing[1] and natural interdecadal variability. Thus, most evidence supports the idea that this hiatus fits well within current knowledge of the climate system and its forcing (Rajaratnam et al., 2015), an important factor being the accumulation of heat in the deep ocean, which has strongly increased in the last decades (Glecker et al., 2016). Historical evidence indicates that pauses and even cooling periods are common in the warming process, and they are followed by strong increases in temperature linked to the release of accumulated heat from the sub-surface ocean (Roberts et al., 2015). This hypothesis would be in line with the record-high global temperature records in 2014 (Mann et al., 2016) and 2015 (NASA, 2016, Figure 1.1). This increase continues strongly in 2016: overall, the 15 highest monthly temperature in the record have all occurred in the 15 months up to July 2016, which mark the longest such streak in NOAA's 137 years of record keeping (NOAA, 2016). 2015 was the first year when the symbolic line of 1ºC warming above pre-industrial levels has been crossed.

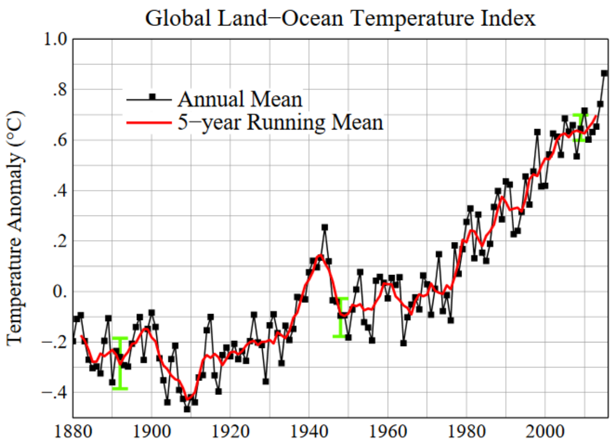

Figure 1.1 Temperature anomaly evolution in the 1880-2015 period, with the base period 1951-1980. Source: NASA (2016)

[1] In climate science, radiative forcing or climate forcing is defined as the difference of insolation absorbed by the Earth and energy radiated back to space (IPCC, 2013)

According to the IPCC (2013) report, **human influence** in the climate system is clear, and the main drivers are anthropogenic emissions of GHG, which are now the highest in history. The main anthropogenic **forcing agents** are emissions of GHG, mainly fossil CO_2 but also biogenic CO_2, methane (CH_4), nitrous oxide (N_2O) and halocarbons. Tropospheric ozone is also a significant positive forcing agent, whereas other agents such as stratospheric ozone and surface albedo are small negative contributors, and stratospheric water vapor from CH_4, contrails, or aerosols-cloud-radiation interactions may represent small, or statistically not significant, contributors to the total anthropogenic radiative forcing. On the other hand, solar irradiance represented a natural negative forcing agent in the 1980-2011 period (IPCC, 2013, Figure 1.2).

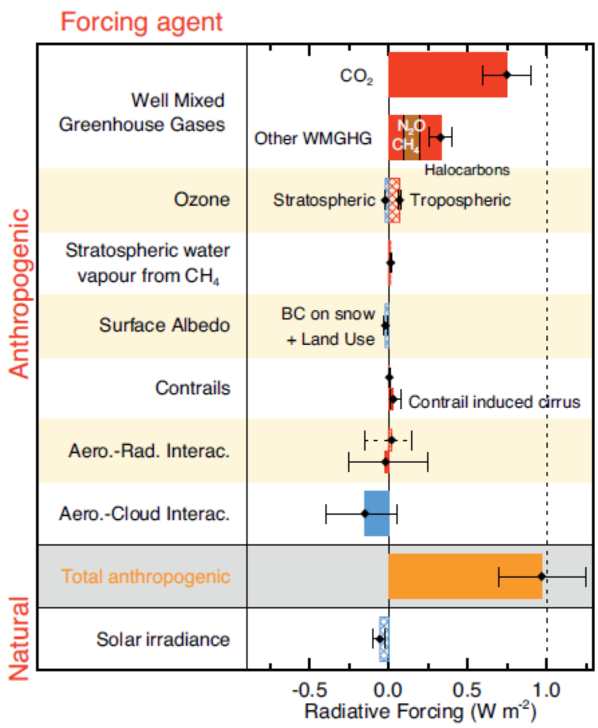

Figure 1.2 Radiative Forcing between 1980 and 2011 (W/m²), per forcing agent. Source: IPCC (2013)

The warming effect of the different GHGs is expressed as the global warming potential (GWP), which refer to the relative radiative forcing per unit mass when compared to carbon dioxide. Due to the different lifetimes of GHGs, the GWP varies with the time period considered. The GWP of CH_4 is much higher for a 20-year horizon (84) than for a 100-year horizon (28) (IPCC, 2013). In its new report, the IPCC (2013) include an additional estimate of the GWP based on the consideration of carbon-climate feedbacks (Gillet and Matthews, 2010). This raises the GWP_{100} of CH_4 up to 34. Figure 1.3 shows the trends from 1970 to 2010 in the total GWP of the main anthropogenic GHG emissions This figure suggests that agriculture is a relatively minor activity with regard to GHG emissions. As discussed in Section 1.2, however, agricultural practices also imply emissions in other activities, particularly biomass burning and all economic sectors implied in the production of agricultural inputs.

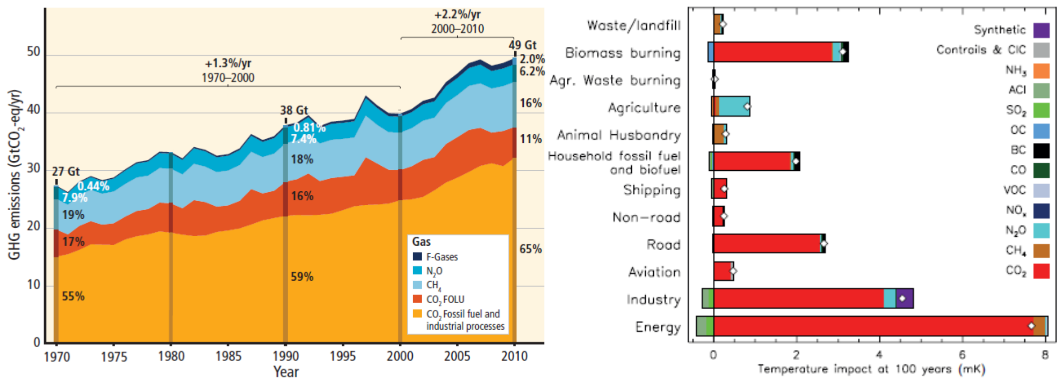

Figure 1.3 Anthropogenic GHG emissions. The evolution of the emissions of the main GHG is shown in a), while b) shows the relative contribution of different human activities to total GHG emissions. Source: IPCC (2013)

Interactions between components of global change: Tackling complexity

The problems related to resource depletion and global change are closely interconnected. The two groups of problems are associated with a linearized economy, which relies on finite resources and finite sinks of residues. The link between the problems suggest that we are facing a single multidimensional crisis. In an interconnected world, this implies complex interactions between the effects of these multiple dimensions including natural and social components.

An example of interaction between components of global change is the relationship between **climate change and biodiversity**. Climate change is expected to have alarming consequences for biodiversity (Bellard et al., 2012), while biodiversity has important ecosystem functions including those related to carbon cycling (see Section 1.2.3). For example, the loss of apex consumers in trophic chains of global ecosystems can have unanticipated impacts on vary diverse processes, such as wildfire, carbon sequestration and biogeochemical cycles (Estes et al., 2011).

An example of **negative feedback** within the climate system is the cloud-phase feedback, through which liquid clouds, more abundant in a warmer environment, would reflect sunlight more effectively than ice clouds. This effect has recently shown to be much less strong than previously thought, implying that equilibrium climate sensitivity (which relates CO_2 concentrations to atmospheric temperature) projected by global climate models could have been underestimated as much as 1.3ºC (Tan et al., 2016).

Positive feedbacks tend to reinforce current changes, and therefore they contribute to change the equilibrium state of the system towards another level. With increasing global temperatures, the risk of onset of positive feedback processes also increases. One of the most known positive feedbacks of the Earth system are the melting of the permafrost, with expected massive methane emissions (Koven et al., 2011, DeConto et al., 2012). The combination of anthropogenic disturbances and positive feedbacks increase the probabilities of major disruptions in the Earth climate systems, such as the collapse of the Atlantic thermohaline circulation (Vellinga and Wood, 2002), and also of crossing "no-return lines", after which the reductions in GHG emissions won't be able to stop the warming process. The maximum cumulative emission which has been considered safe, taking into account the feedbacks, is 500 Pg C, about half of the emissions associated to the commonly established +2ºC maximum global warming target (Hansen et al., 2013).

The indirect impacts of **agriculture** on the climate system are very complex and depend on specific situations. For example, the alteration of the **nitrogen cycle** in Europe, which is largely driven by agriculture, has warming effects through processes such as N_2O emissions and decreased CO_2 sink due to tropospheric O_3, while it has cooling effects through increased CO_2 sink due to N deposition light scattering effects of aerosols, and O_3-driven reduction of the lifetime of CH_4. The overall effect of agricultural-mediated responses to N cycle alterations on the climate system may range from a substantial cooling to a small warming effect (Butterbach-Bahl et al., 2011).

Other important interactions between global change components are those between climate and resource depletion, which are discussed in the following subsection, and those between climate and ecosystem carbon dynamics, which are discussed in Section 1.2.3.

Interactions between climate change and resource depletion

Two opposite trends can be expected from the impact of **peak oil and resource depletion** on GHG emissions, implying that the overall effect will largely depend on societal choices (Kharecha and Hansen, 2008). On the one hand, depletion implies that less fossil fuels are burned each year, and thus their emissions should also decrease. The associated decline in industrial production could also have other positive effects on pollution reduction, such as the decline in urban waste, leading to a "peak waste" phenomenon (Bardi et al., 2014). On the other hand, however, the depletion of easy to extract resources would shift the production towards those which require more energy, or are more carbon intensive but cheaper, and thus cause more emissions per unit energy obtained. Historically, the EROI had grown during the initial phases of oil development due to technological improvements. At some point, however (around 1960 in the world), the importance of the physical component became larger than that of the technological component, and the EROI started declining (Dale et al., 2011). This point could have been around 1960 for world oil and gas (Hall et al., 2014). This trend has become more relevant in the last years, as the share of production and refining in the total carbon footprint of energy sources increases (Aguilera et al., 2015). It is also worth noting that even renewable energy sources face declining EROIs, which would start once metal ore degradation reaches a certain point (Fzaine et al. 2015).

The **intensification in emissions** occurs at multiple levels: conventional oil sources being substituted by unconventional sources such as oil sands, deep water or fracking resources (see a review in Aguilera et al., 2015), gas power generation being substituted by coal, etc. A paradigmatic example is the production of oil sands in Alberta (Canada), which mining and refining processes are very intensive in GHG emissions (Swart and Weaver, 2012), due to the low quality of the resource, which requires a high energy investment to process and refine. But mining also destroys wide areas of boreal forests and peatlands, releasing large amounts of CO_2 to the atmosphere. It is estimated that projected mining projects could release 11-47 Pg of CO_2 to the atmosphere, besides preventing the annual sequestration of 6-7 Tg (Rooney et al., 2012). Another unconventional resource being developed at this moment is shale gas obtained through hydraulic fracture, which can be associated to massive methane leakage from production facilities, with estimations ranging from 3.6%-7.9% (Howarth et al., 2011) to 9.1% (Schneising et al., 2014). At these levels of fugitive emissions, the carbon footprint of natural gas energy would be even higher than that of coal (see Section 1.2.4). Hence, despite the peak of conventional oil occurred in 2005, and renewable energy production is showing a spectacular growth, the concentration of CO_2 in the atmosphere has still kept growing in the last decade. It now surpasses the symbolic value of 400 ppm (Monastersky, 2013), which is 50 ppm above what many scientists consider "safe" (Rockström et al., 2009). On the other hand, climate change is also expected to have impacts on resource depletion dynamics. For example, it will affect international trade by increasing the costs and reducing the reliability of freight transport, potentially resulting in a reversal of globalization trends (Curtis, 2009). The impacts of climate change impose an additional burden to energy production operations. For example, a decrease in precipitation could affect hydroelectricity production, or an increase in surface water temperature could decrease the capacity of water to cool off thermal power plants (Cook et al., 2015, Sanders, 2015).

Interactions of resource depletion and global change with society

Resource depletion and global change are expected to be reflected in severe economic and social impacts. Resource extraction and global change impacts are already related to social conflicts involving local communities affected by the impacts, what have been termed ecological distribution conflicts (Martínez-Alier, 2003, Latorre et al., 2015). The increased resource production effort due to declines in EROIs and mineral ore grades, and the expansion of the extraction frontiers, will surely exacerbate these conflicts (Orta-Martinez and Finer, 2010). The risk of military conflicts over remaining resources, and of the appearance of authoritarian regimes, is also expected to increase due to peak oil (Leder and Shapiro, 2008). Global climate has been shown to have a very strong influence on human conflict (Barnett and Adger, 2007, Hsiang et al., 2011), and ongoing climate change impacts are already increasing conflicts (Hsiang et al., 2013, Schleussner et al., 2016) and poverty (Leichenko and Silva, 2014). Hope and Hope (2013) showed that the impacts of climate change will probably be much more severe in a context of low economic growth, indicating that the need for climate change mitigation is even stronger in a low-growth scenario.

The socio-economic impacts of resource depletion and global change can have multiple environmental consequences, reinforcing or mitigating global change trends. For example, the priority of enviormental considerations may be reduced as energy becomes increasingly scarce (Czucz et al., 2010). Increasing conflicts undermine the governance required to develop and apply regulations. The conflicts generated by global change make the societies more vulnerable to climate variability, because the severity of these impacts is strongly dependent on the socio-political framework. The adaptive capacity of societies is mainly associated with governance, civil and political rights, and literacy (Brooks et al., 2005). Likewise, inequality and environmental impacts are closely related. Climate change exacerbates inequality because the most severe impacts of global change are suffered by the poor, while being caused by the rich. At the same time, inequality has been shown to be an important driver of environmental pollution and degradation (Cushing et al., 2015). For example, Lassaletta et al. (2016) have recently shown how unequal compostion of human diet throughout the world generates an unfair distribution of the environmental consecuences of crop produciton. Animal feed produced for exportation to rich countries is treatining the access of poor people to an equitable diet. On the other hand, countries that produces animal feed are suffering the environmental burdens of the production of traded commodities (Oita et al. 2016).

The environmental crisis, however, could also be used as an opportunity to transform the socio-ecological system into a more desired state, by building up a resilience-focused adaptive governace (Folke et al., 2005). At the local scale, research has shown that reducing inequality could improve environmental sustainability (Fabinyi et al., 2015). Likewise, Steffen and Smith (2013) described very strong synergies between global equity and environmental goals. Overall, assuming that economic degrowth is a necessity imposed by resource depletion and by the need to mitigate global change), there is a need to make it socially sustainable (Kallis et al., 2012)

1.1.2 The impacts resource depletion and climate change on agriculture and food production

Resource depletion can severely impact modern agriculture, which is highly dependent on a reliable supply of energy and material-intensive inputs for its normal functioning (see Section 1.2.4). One impact of diminishing fossil fuel supplies is the increase in the demand for alternative fuels such as **biofuels**. Biofuels proponents argue that they could contribute to energy and environmental goals, having positive EROIs and generating useful coproducts (Farrell et al., 2006). But many other research studies question this view, showing that the net energy balance of biofuels could be very low or even negative (de Castro et al., 2014), implying that cultivation of biofuels is not an energy production process, but rather a process employing one type of energy (such as natural gas in fertilizer production and electricity in irrigation pumps) to produce a different type of energy (liquid fuels). Moreover, the expansion of biofuels imposes a strong demand for agricultural land, promoting deforestation and associated GHG emissions (Fargione et al., 2008). Overall, the net effect of biofuels on climate change would strongly depend on particular conditions such as the amount of land used for feedstock cultivation and the carbon density of this land (Valin et al., 2015).

There is a large uncertainty on the **impacts of climate change** on agriculture, as the variables involved interact non-linearly (Porter and Semenov, 2005). The uncertainty can be largely explained by the interactions between temperature and precipitation changes. Some evidence suggests that, despite the major role of precipitation on the year-to-year yield variability, temperature could have a more important role shaping long-term trends (Lobell et al., 2008), with potentially deleterious impacts due to a higher frequency of extreme temperature events (Battisti and Naylor, 2009). Generally speaking, increases in climate variability could have a more adverse overall effect on crop yields than changes in average climatic trends (Porter and Semenov, 2005). Another important impact of climate change on agricultural resources is its effect on groundwater availability, which adds up to the already existing over-exploitation of groundwater resources in many regions (Green et al., 2011). Climatic change is also projected to affect weed species composition and new species introduction, with major ecological and agronomical implications (Peters et al., 2014). The projected impacts of climate change on agriculture are expected to affect the different world regions unevenly, with usually much more negative impacts in low latitude regions than in high latitude regions, in which the impacts may even be positive (Rosenzweig et al., 2014). On the other hand, another important effect of increasing atmospheric CO_2 concentrations on agriculture is not indirect through climate change but direct through enhancement of photosynthetic activity, what has been called "CO_2 fertilization" (Section 1.2.3). Temperature and CO2 interact in a complex way with plant productivity, resulting in non-linear responses (Dieleman et al. 2012).

1.2 Agriculture as a source or sink of GHG

1.2.1 Quantifying agricultural GHG emissions

Direct agricultural emissions include direct emissions of GHG from soils (mainly N_2O from all soils and CH_4 from flooded land), animals (mainly CH_4 from enteric fermentation) and manure (N_2O and CH_4), crop residue burning emissions (N_2O and CH_4), carbon emissions or sequestration resulting from the soil carbon balance (CO_2), and CO_2 (mainly) from fuel combustion in machinery and heating in the farm. **Indirect agricultural emissions include**, "**upstream**" the cropping system, emissions from the production of agricultural inputs (fuels, electricity, fertilizers, pesticides, machinery and buildings) and "**downstream**" the cropping system, indirect N_2O emissions (mainly from transformation of volatilized NH_3 and leached NO_3^-). We could also include within "downstream emissions" indirect emissions from land use change (Figure 1.4)

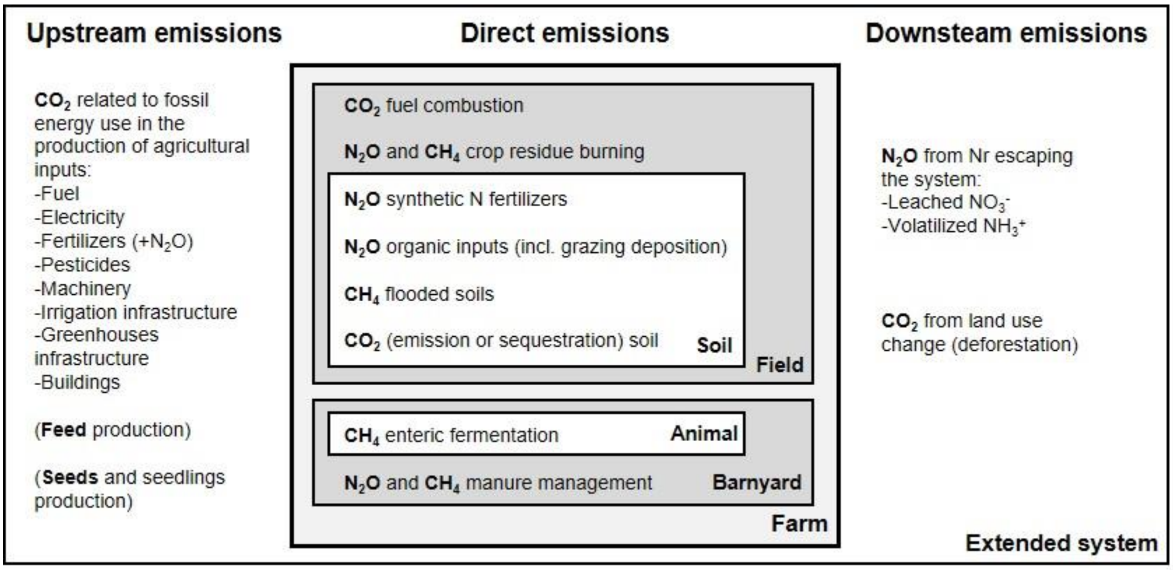

Figure 1.4. Schematic representation of agricultural GHG emissions

Agricultural emissions at the country level are quantified by **National Emission Inventories**, which have to be reported yearly to the United Nations by those countries in the Annex B of the Kyoto protocol. National Inventories commonly represent the most comprehensive characterization of country-based emissions, and are used for the identification of emission hotspots and the design of mitigation policies. Official estimates following IPCC guidelines for National Inventories (IPCC, 1996a, 2006) show a relatively minor role of agriculture and animal husbandry in global GHG emissions (see Section 1.1.1) and particularly in emissions of industrialized societies (e.g. Spain, MAGRAMA, 2013). In spite of this, it is worrisome that global agricultural emissions grew at roughly 1% annually from 2000 to 2010 (Tubiello et al., 2013, 2015), representing 11.2% of total GHG emissions in 2010 (Tubiello et al., 2015). The IPCC categorization, however, only includes specifically agricultural processes, but not processes indirectly required for agricultural activities, such as the industrial production of inputs, which are allocated to other sectors. In the case of cropping systems, they include mainly (with 2010 total agricultural emissions share from Tubiello et al., 2013 indicated in parenthesis): direct and indirect N_2O emissions from soil application of synthetic fertilizer (15%), manure (3%) and crop residues (3%), N_2O and CH_4 emissions from biomass burning (not estimated), and CH_4 emissions from rice paddies (11%). In the case of livestock production, they include N_2O and CH_4 emissions from manure management (8%), N_2O emissions from grazing (17%) and CH_4 emissions from enteric fermentation (44%). Most of the other processes involved in the GHG balance of agricultural systems are classified within the other sectors. For example, fuel combustion emissions are within "Non-road", the emissions associated to the production of industrial agricultural inputs, such as machinery, fuel, while electricity, fertilizers, pesticides or infrastructure are classified within industry, energy and transport. With the expansion of industrialized and globalized agriculture, however, these factors are becoming increasingly relevant. For example, the trade of agricultural commodities has increased 8-fold (when expressed in nitrogen) in the last 50 years (Lassaletta et al., 2014). On the other hand, emissions due to deforestation caused by cropland or grassland expansion (which are the main causes of biomass burning emissions) is included within the Land Use, Land Use Change and Forestry (LULUCF) category.

Therefore, there is a need for **more integrated assessments** of agricultural GHG emissions including the complete account of emissions associated to agricultural activities. As discussed in Section 3.3, life cycle assessment offers a good framework for this task. An estimation of global agricultural emissions using a Life Cycle Assessment (LCA) perspective suggests that since 1970 agricultural emissions could be decoupled from agricultural production: despite production doubled from 1970 to 2010, agricultural GHG emissions peaked in 1991 at 12 Pg CO_2-eq/yr, and remained stable since then, representing ca. 25% of the global GHG burden in 2010 (Bennetzen et al., 2016).

Agricultural emissions can be studied as part of **agri-food system emissions.** Some studies covering the full GHG emission budget of diets have revealed the important role of food consumption in total anthropogenic emissions. For example, Sanfilippo et al. (2012) found that, in a typical working day in urban Europe, the carbon footprint of lunch was often larger than the carbon footprint of transport, depending on lunch composition and commuting distances. Likewise, Grunberg et al. (2010) estimated that the carbon footprint of food production in Germany represented 16-22% of total emissions in the country.

1.2.2 N_2O emissions

The nitrogen cycle

Anthropogenic nitrous oxide emissions from agroecosystems are an unintended byproduct of the transformations undergone by reactive nitrogen along the "nitrogen cascade" (Galloway et al., 1998, 2003). This cascade represents the series of effects of the anthropogenic reactive N (Nr) over the N cycle, describing the movements of the Nr atom among the major compartments of the Earth system (atmosphere, hydrosphere and biosphere). A single Nr atom can undergo multiple transformations and move between different compartments, in a sequence that amplifies the environmental consequences of anthropogenic N fixation (Erisman et al., 2013) until it is finally transformed back to atmospheric N_2 (Figure 1.5).

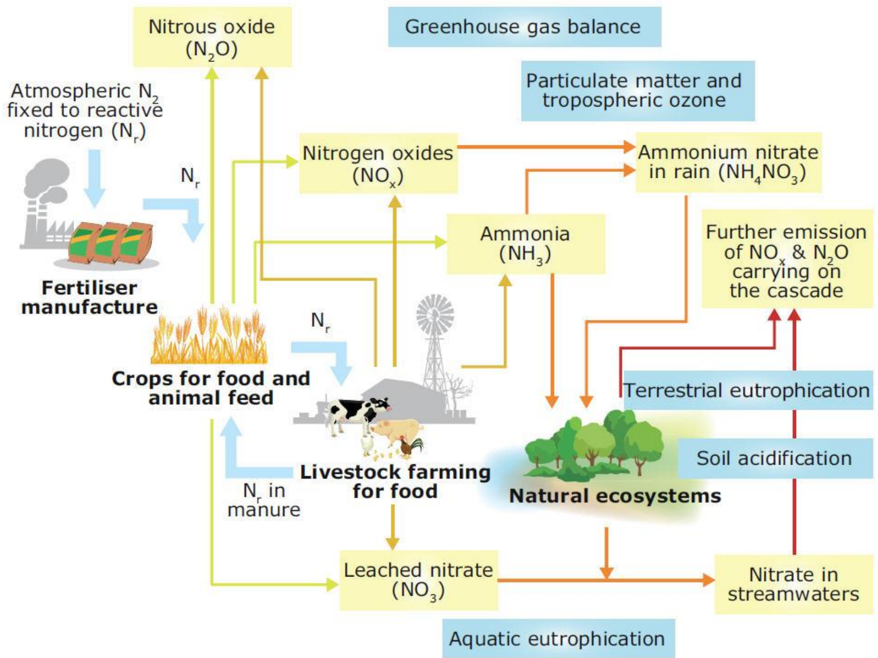

Figure 1.5 Simplified representation of the nitrogen cascade. Source: EEA, 2012, adapted from Sutton et al. (2011)

For most of the anthropogenic N_r, the cascade starts with fertilizer production, where atmospheric N_2 is first used to produce ammonia (NH_3), usually employing natural gas as the source of energy and H atoms. Ammonia is the main feedstock of commercial N fertilizers such as urea and ammonium sulphate, which are applied to cropland soils. Part of this N is uptaken by crops and harvested for food, feed, fiber and bioenergy, but another fraction escapes from the agroecosystem as direct emissions of N_2O and N oxides (NO, NO_2), as volatilized ammonia (NH_3), or as leached nitrate (NO_3). These compounds are transported through the air and water, reaching other agroecosystems and natural ecosystems. Further emissions are produced in the hydrosphere and terrestrial ecosystems when these compounds are transformed: leached nitrate goes to streamwaters, where it emits nitrogen oxides and N_2O (indirect N_2O emissions), while volatilized ammonia and nitrogen oxides are transported and transformed in the atmosphere until being deposited with the air or the rain back to terrestrial and aquatic ecosystems, continuing the Nr cascade. Meanwhile, the N contained in the vegetal proteins of the feed is used by livestock to produce animal proteins, also generating a substantial amount of manure from which can be emitted again as volatilized ammonia, leached nitrate, gaseous emissions. At this stage, N_2O is emitted as direct and indirect emissions from manure management and excreta of grazing animals (Sutton et al., 2013). The N remaining in the manure after its management is applied back to agroecosystem soils, where it continues the cascade. Another important factor shaping the nitrogen cascade is the trade of food and feed products around the globe, which has led to major imbalances in the global distribution of Nr, and its effects (Billen et al., 2013; 2015, Lassaletta et al., 2014a, 2014b, 2016, Oita et al., 2016).

The residence time of Nr in each compartment are very variable, depending on the type of Nr compound and the characteristics of the compartment. These differences lead to accumulation of some forms of Nr in some compartments, intensifying the impacts of the Nr compounds. These impacts can be classified into five main threats (Sutton et al., 2013): i) **water quality**, including dead zones, hypoxia, harmful algal blooms, ammonia, nitrite and nitrate contaminated streams and aquifers; ii) **air quality**, including human health effects of air pollutants such as particulate matter formed from NO_x and NH_3 emissions, and from increased concentration secondary pollutants such as tropospheric O_3; iii) **GHG balance**, mainly N_2O emissions, but also, plus interactions with other Nr forms, or tropospheric O_3. Moreover, N_2O is now the main cause of stratospheric O_3 depletion; iv) **Ecosystems and biodiversity**, including loss of species adapted to low-nutrient conditions. Eutrophication due to Nr deposition is also a threat to biodiversity in protected areas; v) **Soil quality** is negatively affected by over-fertilization and atmospheric Nr deposition, which acidify natural and agricultural soils.

The discovery, in the 18th century, of nitrogen and its importance for living organisms, led to the search for methods to transform atmospheric N_2 into Nr, this is, to break the triple bond of the abundant N_2 molecule (representing 78% of atmosphere composition) in order to obtain molecules that can be directly used by plants, that is, the reactive nitrogen (Smil, 2004, Galloway et al., 2013). In the 1920s, Fritz Haberl and Carl Bosch developed a relatively energy efficient method for the fixation of atmospheric N_2. Since that time, the amount of N fixed by humans have increased exponentially. Natural nitrogen fixation has been estimated to be 63, 140 and 203 Tg N yr^{-1} in the land, oceans and in total, respectively (Fowler et al., 2013). Anthropogenic Nr creation raised from 15 Tg N yr^{-1} in 1860 (Galloway et al., 2008) to 210 Tg N yr^{-1} in 2010 (Fowler et al., 2013), being now similar to natural fixation. The majority, 120 Tg N yr^{-1}, were created by the **Haber-Bosch** process. The amount of N globally fixed by grain legumes has been estimated to be 21 Tg N yr^{-1}, while that of pasture and fodder legumes 12-25 Tg N yr^{-1}, non-legume croplands 10 Tg N yr^{-1} and extensive savannas 14 Tg N yr^{-1}, summing up 50-70 Tg N yr^{-1} fixed biologically in agricultural systems (Herridge et al., 2008). NO_x emissions from fossil fuel combustion are another important source of anthropogenic Nr creation, representing 30 Tg N yr^{-1} in 2005 (Galloway et al., 2008).

Global N_2O emissions

Atmospheric concentrations of N_2O have risen from a very stable level in preindustrial times, around 270 ppb, to 319 ppb in 2005 and 321 ppb in 2011 (Robertson and Vitousek, 2009, Myhre et al., 2013), and they are still increasing fast. In 2011, emissions of N_2O represented 6% of the total anthropogenic radiative forcing, being the third most important gas after CO_2 and CH_4. Before ca. 1850, anthropogenic N_2O emissions were mostly due to agriculture, and they were compensated by lower natural N_2O emissions due to cropland expansion. From 1850, the net additions to the atmosphere started to be positive and atmospheric N_2O concentration started growing (Kroeze et al., 1999). By the end of the 20th century, anthropogenic N_2O emissions represented about 44% of total N_2O emissions (Barton and Atwater, 2002).

Global N_2O **emissions from soils** have been estimated to increase from 8.8 Tg N yr-1 in 1900 to 11.3 Tg N yr-1 in 2000 (Bouwman et al., 2013b). Another study estimated anthropogenic N_2O emissions from soil to be 3.3 Tg N yr^{-1}, while they were 1.5 Tg N in rivers and estuaries (Galloway et al., 2004). According to this study, the sum of agricultural and livestock emissions would represent 60% of anthropogenic N_2O emissions, although other estimations suggest that livestock alone (including soil emissions from grasslands and cropland for feed) could represent 65% of the anthropogenic N_2O budget (Steinfeld and Wassenaar, 2007).

Nitrous oxide is usually the dominant component of the **GHG emission balance of cropping systems**. It represented more than half of agricultural emissions in the US (Snyder et al., 2009). And more exhaustive LCA studies at the farm scale also confirm this pattern. For example, Robertson et al. (2000) found that N_2O was the main GHG in three out of four cropping systems studied in the American Mid-West. Similar results were obtained by Adler et al. (2007) in a LCA study of biofuel feedstock production using DAYCENT model to estimate GHG emissions. In Germany, Flessa et al. (2002a) found that N_2O represented about 60% of the global warming potential of organic and conventional mixed farming systems.

Processes involved in direct N_2O emissions from soils

Much of the uncertainty in the assessment of the carbon footprint of crop production is related to the quantification of N_2O emissions (Del Grosso et al., 2014). At the same time, improved crop production has been identified as the major strategy to reduce N_2O emissions (Oenema et al., 2014). Most N_2O emissions are the result of transformations of fertilizer N (either synthetic or organic). It has been estimated that ca. 2% of manure N and 2.5% of synthetic N has been emitted as N_2O since 1860 (Davidson, 2009). A fraction of these emissions (about 1% of N applied on average) arise from direct transformations of fertilizer N in soils, and they are named "direct emissions", while the rest are indirect emissions arising from the transformation of other N_r compounds escaping the farm. The Nr generating direct N_2O emissions can not only come from synthetic and organic N inputs to the soil, but also from the mineralization of soil organic matter. IPCC (2006) recommends calculating N_2O emissions from soil organic matter mineralization when a net loss of organic matter is taking place in the soil.

Nitrous oxide emissions from agricultural soils are mainly the result of **nitrification** and **denitrification** processes mediated by microorganisms (Firestone et al., 1980, Firestone and Davidson, 1989). Nitrification is the microbial oxidation of NH_3 into NO_3^-, and denitrification is the microbial reduction of NO_3^- into N_2 (Figure 1.6) Nitrification occurs under aerobic conditions and denitrification occurs under total or partial anaerobic conditions.

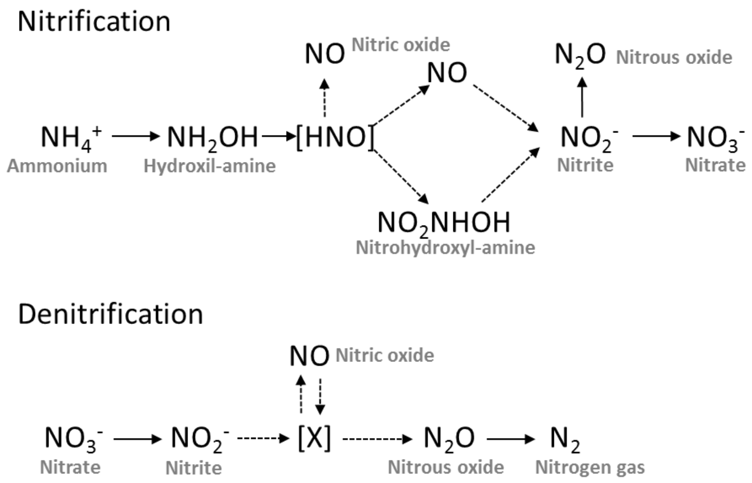

Figure 1.6 Schematic representation of nitrification and denitrification pathways. Source: Own elaboration from Barton and Atwater (2002), Hooper (1984), Firestone and Davidson (1989)

Nitrous oxide is an intermediate molecule in denitrification and an unintended byproduct of nitrification, and in both processes a portion of it escapes from microorganisms to the soil and the atmosphere. Nitrification is favored by low water contents in the soil, which allow aerobic conditions, while denitrification benefits from the opposite conditions. A third process for N_2O production in the soil is nitrifier denitrification, which is the pathway of nitrification, carried out by autotrophic nitrifiers, in which NH_3 is oxidized to NO_2- followed by the reduction of NO_2- to

NO, nitrous oxide N_2O and N_2. This pathway might represent 30% of N_2O production in soils, and is favored by low oxygen coupled with low soil organic carbon (Wrage et al., 2001, Kool et al., 2010, 2011).

Some of the N_2O generated in nitrification, denitrification and nitrifier denitrification is consumed by soil microorganisms. Actually, the soil can act as a sink of N_2O, when N_2O consumption is larger than N_2O production, which occurs quite frequently although usually only very transiently (Chapuis-Lardy et al., 2007). N_2O consumption in natural ecosystems, mainly peatlands and wetlands, represents about 2% of anthropogenic N_2O emissions (Schlesinger 2013).

Factors influencing direct N_2O emissions from soils

N_2O production in the soil is influenced by climatic factors, soil properties, interactions between N transformations in soils and plants, and management practices, leading to a high variability in direct N_2O emissions (Mosier et al., 1996, Chen et al., 2008). The most usual metric to assess the impact of these variables on N_2O emissions relates them to N fertilizer inputs and is called N_2O emission factor (EF). The EF can be defined as the share of input N that is emitted as N-N_2O, discounting "background" emissions, which are measured as emissions in an unfertilized control treatment.

The most important influencing factors are soil humidity (measured as Water Filled Pore Space, WFPS), temperature and nitrogen availability. In turn, these factors can be affected by other drivers such as soil properties (e.g. clay content, organic matter content) climate patterns, fertilizer type or management practices (e.g. irrigation, tillage)

Temperature is a direct driver of microbial activity, and consequently also of N_2O emissions. The N_2O EF usually increases with temperature, interacting with WFPS (Flechard et al., 2007).

WFPS determines soil aeration, and thus its redox potential, conditioning the biochemical reactions that can be performed by soil microorganisms. It also imposes limits to microbial activity and survival (i.e. too dry conditions). Nitrification dominates in the 20%-60% WFPS range, while denitrification dominates in the 50%-100% range and becomes increasingly efficient in beyond 70% (Vilain et al., 2010), reducing NO_3^- all the way to N_2, and thus decreasing N_2O emission under very humid conditions. The evolution of the soil humidity pattern along the year also influences N_2O emissions through its influence on organic matter mineralization processes and substrate availability. Wet-dry cycles promote soil organic matter mineralization, NO_3^- accumulation in dry periods and N_2O emissions in wet periods (Dalal et al., 2003). In seasonally dry soils, emissions can be negligible during dry periods but show large pulses when they are interrupted by rain or irrigation (Dick et al., 2001). Soil **compaction** also affects soil aeration, and thus N_2O emissions. Soil N_2O emissions after a dry period could be 20-fold higher in a compacted soil versus an uncompacted one (Beare et al., 2009).

Nitrous oxide emissions usually respond non-linearly to the **amount of N applied**. Thus, the N_2O EF would be lower at low N rates and higher at high N application rates, this is, it would be better represented by an exponential relationship instead of a linear relationship between N applications and emissions of N_2O (Hoben et al., 2011, Shcherbak et al., 2014, Philibert et al., 2012a, Gerber et al., 2016). Moreover, many authors have called for the study of N_2O emissions related to yield output instead of fertilizer input (yield-scaled emissions) (e.g. Van-Groenigen et al., 2010, Sanz-Cobena et al., 2012, Skinner et al., 2014, Plaza-Bonilla et al., 2014).

Fertilizer type is another variability source in N_2O emissions. Despite the IPCC (2006) does not differentiate EFs by fertilizer type, suggesting that, on a world average basis, the differences between them would be nonsignificant, some works have shown that some differences may exist on specific conditions. For example, some researchers have found that N_2O emissions might be influenced by the type of synthetic N fertilizer (Gagnon et al., 2011) or crop residue (Novoa and Tejeda, 2006). Moreover, fertilizer additives such as nitrification and urease inhibitors can also affect N_2O emissions (Section 1.3).

Other factors affecting soil N_2O emissions are pH, texture, salinity and limitation of nutrients other than N. Soil **pH** affects nitrification and decreases N_2O/N_2 ratio. Soil **texture** affects water retention and aeration, with finer textures favoring denitrification and potentially increasing N_2O emissions (Del Grosso et al., 2006), although different reviews have shown contradictory results (Stehfest and Bouwman, 2006, Novoa and Tejeda, 2006). **Salinity** inhibits both nitrification and denitrification, but it affects N_2O reductase first, thus increasing the N_2O/N_2 ratio. A **deficit of nutrients** other than N can limit N absorption by plants, thus increasing soil N concentration and N_2O emissions (Dalal et al., 2003).

Examples of management practices that can influence N_2O emissions are pesticides, tillage and irrigation. Application of **pesticides** can affect N_2O emissions through interaction with the metabolism soil microorganisms, with effects that might vary depending on the type of pesticide and pedoclimatic conditions (Tenuta et al., 1996, Kinney, 2005, Spokas et al., 2006). **Tillage** also seems to affect N_2O emissions differently depending on pedoclimatic conditions. Antle and Ogle (2012) estimated that N_2O emissions would decrease after adoption of no tillage in the western US estates, but they would increase in the eastern estates due to distinct pedoclimatic conditions. **Irrigation** has a clear impact on soil biochemical processes through its influence on WFPS, which would depend on the type of irrigation technology employed, such as flooding, furrow, sprinkler, drip, or subsurface drip irrigation. High-humidity conditions favoring denitrification are expected to lead to highest N_2O emissions (Kennedy et al., 2013). Thus, water-saving techniques are usually associated with lower N_2O emissions than water-intensive irrigation techniques (Liu et al., 2011), which is supported by studies comparing drip irrigation and furrow irrigation (Wang et al., 2016, Zhang et al., 2016), an effect that could be more pronounced with sub-surface irrigation systems (Maris et al., 2015). The effect of irrigation on N_2O emissions is short-termed, mainly through its direct effect on WFPS. This is in accordance with studies finding no effect of irrigation on emissions in the non-growing (and non-irrigated) season (Hao, 2015). **Plastic mulch** covers could also have an effect on N_2O emissions, although this may vary widely depending on site-specific agro-climatic conditions, including significantly decreasing (Berger et al., 2013), having no effect (Liu et al., 2014a) or significantly increasing N_2O emissions, particularly under high temperature and humidity conditions associated to solarization practices (Arriaga et al., 2012, Nishimura et al., 2012). Last, N inputs from depositions of **grazing** animals has a higher emission factor (2%) than N from fertilizers (1%) (IPCC, 2006). Some evidence suggests, however, that the effect of grazing on N_2O emissions at the ecosystem scale may not always be positive, for example in cool semi-arid steppes (Wolf et al., 2010).

Indirect N_2O emission

Indirect N_2O emissions are generated when surplus Nr compounds (surplus = N applied – N extracted in the harvest) volatilized or leached from the farm are transformed outside of it. They take place in in rivers, estuaries, other water bodies and soils of natural and managed ecosystems (Garnier et al., 2009, Vilain et al., 2012). In soils of agroecosystems, these emissions are part of the "background emissions" measured in unfertilized "control" plots, and arise from the transformations of the N coming from other agroecosystems and reaching the soil via deposition or irrigation. There is little information about N_2O emissions in estuaries: the available data suggest that they relatively small but highly influenced by global changes such as species composition, NO_3^- concentrations and oxygen concentrations (Murray et al., 2015). Recent evidence suggests that IPCC estimation of indirect N_2O emissions from rivers could be underestimated by nearly one order of magnitude, due to the exclusion of zero-order streams, where N_2O fluxes are greatest (Turner et al., 2015). This finding would help explaining the mismatch between top-down ad bottom-up estimations of N_2O emissions (Griffis et al., 2013).

1.2.3 Carbon sequestration

Concept and history

The term **soil organic matter (SOM)** is today broadly employed to name the non-living organic product of the decomposition of animal, vegetal, bacterial and fungal tissues. It is widely recognized as an indicator of soil quality, due to its major role in soil fertility. SOM has multiple functions in the soil, including increased water holding capacity, increased nutrient retention capacity, improvement of structure and soil biological quality (Carter, 2002, Diacono and Montemurro, 2010), with additional benefits such as the reduction of erosion, the increase in biodiversity, and the suppression of pests (Akhtar and Malik, 2000, Ghorbani et al., 2008). These characteristics also allow buffering climate impacts on agronomic performance Thus, the loss of SOM is usually associated to soil degradation, while SOM increase stands as a major strategy to adapt to climate change (Lal, 2010).

The concept of **soil organic carbon (SOC)** refers to the C contained in SOM. A standard carbon content in SOM of 58% (Mann, 1986) is widely accepted in the literature to estimate SOC from SOM information or vice-versa. SOC represents an important global carbon pool, containing twice as much carbon as the atmosphere and three times more than the living biomass. Soil inorganic carbon (SIC) is also an important global carbon pool, representing a similar amount as the atmosphere (Lal, 2004a). Through **soil carbon sequestration**, a fraction of the CO_2 captured by plants through photosynthesis is stored in the soil, contributing to increase the SOC pool. Carbon sequestration has been identified as the agricultural process with the highest mitigation potential (IPCC, 2007a, Smith et al., 2008a). To illustrate this potential, Lal (2010) calculated that growing vegetation can photosynthetize as much as 400-fold the increase of CO_2 taking place on the whole air column above it. The total mitigation potential of C sequestration in agricultural soils may range between 0.4 and 1.2 Pg C/yr for 20-50 years (Lal, 2004b) while the total anthropogenic C emissions were 13.4 Pg C in 2010 (IPCC, 2013).

There are **limitations to the potential of carbon sequestration as a mitigation strategy**. Three major ones were outlined by Powlson et al. (2011): i) The quantity of carbon that can be stored in the soil is finite; ii) the process is reversible, and iii) it can lead to increases in the fluxes of other GHG. The issue of the limited storage (Six et al., 2002) is important mainly from a dynamic perspective: the "mitigation effect" of carbon sequestration would only take place until a new equilibrium is achieved. Another important aspect for the quantification of GHG mitigation through carbon sequestration is the origin and alternative use of the sequestered carbon: only additional storage of carbon should be considered sequestration (Powlson et al., 2011). On the other hand, Sommer and Bossio (2014) estimated the effect of a gradual implementation of sequestration measures and of their temporal limitation, concluding that global SOC sequestration has a low climate change mitigation potential and cannot be considered as a climate stabilization wedge, as defined by Paccala and Socolow (2004). Their conclusions, however, were revisited by Lassaletta and Aguilera (2015), who showed that carbon sequestration actually met the criteria for being considered as a carbon stabilization and this mitigation strategy is still very promising particularly for the next crucial decades.

The carbon cycle and its interactions with climate change

Major changes have occurred in the global carbon cycle during the last century. Land use changes have led to the emission of 129 Pg C to the atmosphere, while 98 Pg C were offset by net uptake due to increased CO_2 concentration and climatic changes, resulting in a net release from terrestrial ecosystems to the atmosphere of 31 Pg C (Piao et al., 2009).

The **coupling of climate and the carbon cycle** (IPCC, 2007b) is a core process in climate dynamics. The increase in CO_2 concentration affects climate, which alters the processes responsible for the fluxes of CO_2 to the atmosphere. The relationship between both systems is complex, and it is comprised of multiple interactions, some of which are negative (they tend to stabilize the system) and other positive (they tend to accelerate the change). A major example of negative feedback is the effect of the atmospheric CO_2 concentration on plant growth: higher CO_2 concentrations promote plant growth, which promotes carbon sinks in biomass (Curtis et al., 1998) and the soil (Jain et al., 2005), decreasing atmospheric CO_2 concentration. The enhancement of plant growth by increased CO_2 might be amplified by changes in plant species (Polley et al., 2012). This **CO_2 fertilization** effect is not expected to be able to reverse the negative effects of climate change on agricultural production, particularly in the most vulnerable areas (Müller et al., 2015). Recent evidence, however, suggests that drought stress reduction driven by increased CO_2 (due to reduced stomatal conductance and plant water use) may be underestimated in Earth system models (Swann et al., 2016).

Land and ocean are the two major carbon sinks in the Earth system, although their CO_2 uptake rate is declining in the last decades (Raupach et al., 2014). The increase in primary production could indirectly promote carbon sequestration in the soil, although this process might be overhauled by other interactions (Schlesinger and Andrews, 2000). In particular, there is a positive feedback between temperature and **soil respiration**, which would increase the release of carbon from the soil (Davidson and Janssens, 2006, Smith et al., 2008b, Hopkins et al., 2012). But CO_2 effects on photosynthesis and temperature effects on soil respiration are not independent processes, and they are not the only ones influencing carbon dynamics. **Other feedback loops** between climate and soils are interactions between microbial metabolism and permafrost dynamics, the microbial "priming effect", by which old soil carbon degradation is enhanced by new carbon inputs, and the interactions between C and N cycles (Heimann and Reichstein, 2008). Overall, the land-biosphere system represents an increasingly positive feedback to anthropogenic climate change, which has been estimated to amplify climate sensitivity by 22-27% (Stocker et al., 2013).

Multiple studies suggest that **nitrogen and other nutrients** play an important role mediating carbon responses to climate change, and they are also affected by them in a non-linear way (Manzoni and Porporato, 2007, Bonan, 2008, Solokov et al., 2008, Hungate et al., 2009). In particular, nutrient limitation of net primary productivity is an important factor in the carbon cycle (Wieder et al., 2015). Thus, the release of large quantities of N and P to the biosphere has been related to a carbon sink through the removal of nutrient limitation and subsequent increase of photosynthetic activity (Mackenzie et al., 2001, Fisher et al 2012). In fact, the N cycle might interact with the C cycle in many ways, sometimes enhancing sinks or sources, and other attenuating them (Jain et al., 2009). The overall effect of N enrichment on the C cycle seems to have been an enhancement of the carbon sink: anthropogenic N additions since 1860 were estimated to have enriched the terrestrial biosphere by 1.3 Pg N, supporting the sequestration of 11.2 Pg C (Zaehle, 2013). Other evidence, however, suggest that belowground biomass production would not increase in response to increased N inputs, diminishing the response of soil C stocks to N inputs (Liu et al., 2010). On the other hand, atmospheric CO_2 also promotes N sequestration: CO_2 fertilization increased terrestrial carbon storage by 134.0 Pg C, which increased the terrestrial nitrogen stock by 1.2 Pg N (Zaehle, 2013). The C-N interaction would be stronger in disturbed ecosystems than in those near to equilibrium (Gerber et al., 2010). Another example of negative feedback is the enhancement of biogenic aerosol formation, which diminish warming absorbing radiation and facilitating cloud formation (Paasonen et al., 2013).

Factors influencing SOC dynamics

SOC content is the result of a balance between inputs of C to the soil, both internal and external, and outputs from the soil, mainly through soil respiration (organic matter mineralization) and erosion. Therefore, all factors affecting both C inputs and outputs are potentially involved in C sequestration (Figure 1.7), making agricultural management the major variable explaining changes in regional carbon stocks (van Wesemael et al., 2010).

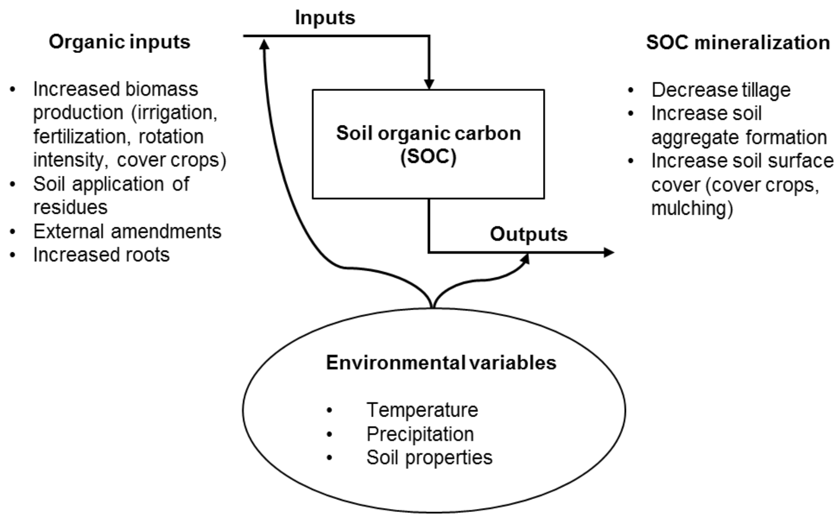

Figure 1.7. Simplified representation of carbon dynamics in agricultural soils, and practices aimed to increase SOC. Own elaboration based on West and Six (2007)

SOC dynamics in agroecosystems are very complex and, despite there has been a large research effort to understand stabilization mechanisms, this is still a highly controversial subject. A central role of **aggregates** (organo-mineral complexes) in SOC stabilization and soil properties is widely recognized after a century of research on the link between soil biotic activity, SOM dynamics and aggregate dynamics (Six et al., 2004, Six and Paustian, 2014). The research on soil aggregate dynamics, however, was still very heterogeneous and a comprehensive theory was lacking by the end of the 20th century (Amezketa, 1999). Sollins et al. (1996) proposed a model in which stability of SOM depended on **recalcitrance** of the raw organic material, **interactions** between soil molecules (organic and inorganic) and **accessibility** of the organic substances to microbes and enzimes, while Six et al. (2002) classified SOM pools in one fraction being physically stabilized, another in intimate association with silt and clay particles, and another biochemically protected in recalcitrant SOM compounds. Von Luetzow et al. (2007) systematized SOM fractioning methods, identifying the difficulty of linking these fractions to specific turnover rates. On the other hand, the understanding of the role of microbial processes in soil dynamics has grown in the last years (Six et al., 2006, Kuzyakov, 2010, Kallenbach et al., 2015), while the link between biochemical properties of organic materials and specific turnover rates has proven to be difficult to stablish (Dungait et al., 2012, Castellano et al., 2015). Indeed, increasing evidence suggests that environmental and biological controls predominate over SOM structure to determine its stability in the soil (Schmidt et al., 2011). Many of these scientific findings have not been incorporated to SOM models, compromising their reliability (Campbell and Paustian, 2015).

Environmental variables influencing SOC dynamics include temperature, precipitation and soil properties. **Temperature** is an important driving factor of microbial activity, and thus on SOC respiration. **Precipitation** effect on SOC is mainly indirect, through increase of NPP, but it also has a direct effect enabling biological activity in the soil, and thus SOC mineralization (Zhang et al., 2013), as also occurs with irrigation (see below). SOC is usually positively correlated to precipitation in natural and managed ecosystems (Doblas-Miranda et al., 2013, Meersmans et al., 2012).

Despite **soil types** seem bad predictors of soil carbon turnover rates (Simfukwe et al., 2011), some soil properties clearly affect SOC dynamics, particularly texture and pH. Soils with coarse **textures** have more aeration, favoring soil respiration, and a lower clay content hinders the formation of aggregates, exposing the soil to mineralization (Melero et al., 2007). Thus, coarse-textured soils are usually associated to lower SOC levels than fine-textured soils (Reijneveld et al., 2009). Moreover, negative charges of clay particles facilitate the attachment of organic molecules, forming aggregates protected from microbial degradation which allow the storage of higher quantities of SOC (McGiffen et al., 2004). As a result, clay content has been identified as the most important variable in regulating the decomposition of SOC (Xu et al., 2016). Low **pH** hinders organic matter mineralization in the soil, potentially leading to higher SOC contents in acidic soils than in neutral or basic ones (Calvo de Anta et al., 2015).

Agricultural practices disturb soils and ecosystem functioning, typically decreasing SOC stocks (Lal, 2004b), particularly in intensively cultivated systems (Zinn et al., 2005).

Roots are a very important source of C to the soil, due to the high quantity of C they represent and the high proportion of their C that is retained in the soil (e.g. Puget and Drinkwater, 2001), particularly of refractory carbon (Katterer et al., 2011). In addition, roots transfer large amounts of C to the soil through rhizodeposition (Johnson et al., 2006). The result of the combination of all those variables is that most soil carbon could be root carbon (Rasse et al., 2005). Root growth is affected by management: carbon allocation to roots is typically larger in poorer environments, such as soils with lower nutrient availability due to organic instead of mineral fertilization (Chirinda et al., 2012), or rainfed soils as compared to irrigated soils (Campbell and de Jong, 2001). Moreover, a more extensive root system in perennial herbaceous plants is responsible for most of the observed differences in SOC stocks of grassland and cropland (Conant et al., 2001).

Crop residues have been considered "agriculture largest harvest" (Smil, 1999). Residue removal has been associated to a series of deleterious effects on soil quality, including physical, chemical and biological properties (Blanco-Canqui and Lal, 2009). Restituting crop residues to the soil can avoid SOC losses associated to residue extraction (Buysse et al., 2013). Burning residues is usually associated to SOC depletion (e.g. Wand and Dalal, 2006), but it could also cause no effect on SOC (Virto et al., 2007) or even promote C sequestration due to the high stability of the remaining C (Knicker et al., 2012). The type of crop residues also influences the stability of carbon in the soil. Legume residues, with a low C:N and lignin:N ratios, have been associated with more efficient conversion of their biomass to stable SOM (Carranca et al., 2009). For example, Bonciarelly et al. (2016) found higher C sequestration rates in a legume-faba bean rotation than in a wheat monoculture rotation, despite higher residue C inputs in the second. Likewise, the inclusion of legumes in cover crops has been associated to increased C sequestration (Conceicao et al., 2013). A negative relationship between C:N ratio and C sequestration rate has also been found for compost amendments (Ryals et al., 2015).

External organic inputs such as manure and compost have been proved as effective strategies to increase soil carbon content (Buysse et al., 2013), and the percentage of their carbon retained in the soil is typically larger than that of crop residues (Triberti et al., 2008, Bertora et al., 2009), although their net effect on carbon sequestration would depend on the alternative fate of the material (Powlson et al., 2011, Gattinger et al., 2012).

Inorganic fertilizers, particularly N, promote plat growth, and thus potentially higher C inputs to the soil, but they usually decrease root to shoot ratio (Poorter and Nagel, 2000, Wang and Taub, 2010) and they may either increase (Geisseler et al., 2014) or decrease (Liu and Greaver,

2010) soil microbial biomass. The net effect on soil respiration is neither clear. A positive response is usually found (Lu et al., 2011) although it may have the opposite effect (impeding SOM decomposition) (Liu and Greaver, 2010), particularly in certain situations such as temperate forest soils not limited by N (Janssens et al., 2010). Many authors have found higher SOC content in fertilized plots than in unfertilized plots (e.g. Blanco-Canqui and Schlegel, 2013, Lu et al., 2011), but there has also been found a very small effect (Liu et al., 2009), an absence of effect (Banger et al., 2009, Lu et al., 2011), and even a depletion of soil carbon after long-term N fertilization (Khan et al., 2007, Mulvaney et al., 2009).

It is also important to take into account the **stoichiometry** of elements in soil organic matter and applied organic and inorganic inputs. The stoichiometry of the stable fractions of soil organic matter is fairly stable. If we take as an example the C:N:P ratio of this fraction measured by Kirkby et al. (2013), averaging 60:5:1, we can infer that per Mg SOC sequestered, 83 kg N and 17 kg P have to be sequestered as well. This implies that, on the one hand, carbon sequestration in stable pools is dependent on nutrient availability and, on the other hand, some of the nutrients will remain sequestered in SOM for a long period before being available to plants (Kirkby et al., 2013, 2014), which means that the positive effect of these practices on soil fertility (apart from intrinsic benefits of higher SOM levels, see Section 5.3) might be delayed until the SOM approaches equilibrium, thus releasing as much nutrients as it gains.

The net effect of **soil erosion** on the carbon cycle is still subject to controversy. Soil erosion removes preferentially the topsoil layer, which is usually the one with the highest C contents. This soil is then exposed to air and thus is more vulnerable to mineralization, potentially making erosion a major source of C emissions (Lal, 2003). However, a fraction of it gets buried in colluvial soils and sediments of water bodies, potentially being stored there for a long time (Stallard, 1998, Wang et al., 2014), which is usually longer for underwater sediments than terrestrial ones (Van Oost et al., 2012). Therefore, the net effect of soil erosion may range from a net source to a net C sink, ranging from -3.4 to 1.1 Pg C/yr (Bilings et al. 2010), although isotope-based assessments suggest that the overall magnitude might be much lower: a net sink of 0.06-0.27 Pg C/yr (Van Oost et al., 2007). Whatever its direct effect on the C cycle would be, erosion has a clear deleterious effect on soil quality where it is produced, potentially leading to yield reductions (de la Rosa, 2000).

Irrigation increases land productivity, which is associated to higher available C inputs to the soil. On the other hand, irrigation also reduces C allocation to roots and affects SOC dynamics by improving conditions for microbial growth, thus promoting soil respiration. The net balance would mainly depend on the relative dominance of one or the other process (higher C inputs versus higher mineralization), and could potentially be improved by increasing the return rate of additional crop residues. Experimental studies have found both higher (Wu et al., 2008) and lower (Nunes et al., 2007, Martiniello, 2011) SOC stocks under irrigation conditions.

Tillage operations disturb the soil, promoting mineralization and a redistribution of organic matter within the soil. They also facilitate the contact between soil particles and crop residues. Net effects of tillage on SOC have been the subject of an intense debate within the scientific community in the last two decades. By the turn of the century, there was a consensus supporting the idea of a clear effect of no tillage promoting C sequestration (e.g. Post and Kwon, 2000, Six et al., 2004, Lal, 2004a, 2004c). During the following years, however, some researchers called attention on methodological issues questioning those conclusions. No tillage promotes soil stratification (Franzluebbers, 2002), with an important SOC accumulation on top soil layers. A substantial number of systematic reviews showed that SOC content could be lower in deep layers, even below tillage depth, which could offset C gains in top layers (Manlay et al., 2005, Baker et al., 2007, Angers and Eriksen-Hamel, 2008, Luo et al., 2010, Powlson et al., 2014, Alcantara et al. 2016). This implies that the conclusions of many previous studies reporting higher SOC contents under no tillage could have been biased by low sampling depth. Another important methodological issue in studies of tillage systems is the difference in carbon inputs among treatments. In a meta-analysis of published data, Virto et al. (2012) found that the differences in SOC content between tillage treatments could be explained by differences in carbon inputs. An additional factor involved in the effect of tillage on SOC dynamics is photodegradation, an important process in semi-arid systems (Henry et al., 2008, Rutledge et al., 2010) that cause the release of residue carbon to the atmosphere without cycling through the soil and could also led to trace GHG emissions (Lee et al., 2012).

Liming has multiple effects on SOC, including promoting biological activity (and consequently SOC mineralization), contributing to SOC stabilization, and increasing C inputs through enhanced plant growth. The combined effect of these processes is frequently an increase in SOC stocks, but reductions have also been observed (Paradelo et al., 2015).

Grazing crop residues can affect SOC stock in multiple ways depending on specific conditions. The net effects observed range from negative (Ryan et al., 2008, Sainju et al., 2010) to neutral (Quiroga et al., 2009) and positive (Hatfield et al., 2007). Grazing of grasslands, if it is moderate, is usually associated to SOC increases (Silver et al., 2010, Ziter et al., 2013), while the intensification of grassland has been associated to SOC depletion in deep soil layers (Ward et al., 2016).

Other pools of carbon in agroecosystems

Soil inorganic carbon (SIC) is usually in the form of calcium carbonate ($CaCO_3$) and dominates soil carbon stocks in many soil types. $CaCO_3$ usually precipitates in the soil from dissolved CO_2 if the conditions are appropriate. Dissolved carbon from root exudates can precipitate as carbonates and be sequestered in that form (Manning, 2008). Soil inorganic carbon can be released as CO_2 due to soil acidification (Yang et al., 2012). The effect of irrigation on SIC can vary depending on water and soil properties (Wu et al., 2008). Carbonate can dissolve and be exported from the soil with the overuse of N fertilizers (Moreno et al., 2006). Eshel et al. (2007) and Díaz-Hernández (2010) have drawn attention on the importance of studying deep soil layers and subsoil when measuring SIC stocks. The dissolution of SIC from top soil layers and re-precipitation in deep layers is a common process that could bias estimations of SIC changes if measurements are not deep enough.

Litter can represent significant amounts of carbon in certain agricultural systems such as organic system (Marinari et al., 2007). Litter dynamics depend on management: fertilization and irrigation promote NPP and thus debris production, but they also accelerate debris mineralization, which could potentially offset the increase in debris inputs (Kochsiek et al., 2009).

Woody biomass is accumulated by woody crops during their growth. Some tree crop cycles, such as those of olives (with the exception of super-intensive plantations) are very long, implying a long-term sequestration of woody biomass until the plantation is renewed. As with the other carbon pools, carbon accumulation in woody biomass is only considered net mitigation when there is a growth in the stock of carbon. This can occur due to an expansion of woody crop surface are, or to an increase in the carbon density of woody crops (for example, due to increases in tree density). Important C accumulation in woody crops biomass has been observed in Mediterranean areas such as California (Kroodsma and Field, 2006) and Andalusia, Spain (Muñoz-Rojas et al., 2011).

1.2.4 Emissions from the production of inputs and energy use

Full GHG accounting of cropping systems should include not only direct and indirect soil emissions but also emissions associated to the production and use of agricultural inputs, which are associated to a significant carbon footprint (Lal, 2004b). The carbon footprint of agricultural inputs would depend on the amount of inputs employed and the carbon intensity of their production, which is largely correlated with its energy efficiency. The use of external inputs in agriculture increased from being almost absent in traditional agricultural systems, which relied mainly on on-farm energy and material fluxes, to representing a significant share of the total inputs consumed (Guzman and Gonzalez de Molina, 2015). This intensification trend was still observed in many countries in the 1991-2003 period (Arizpe et al., 2011). On the other hand, major energy efficiency gains (and consequently carbon footprint drops) were achieved in industrial processes involved in the production of agricultural inputs during the 20th century. In spite of this, we are now in a context of a global economy approaching its limits, with declining EROIs of fossil fuels (Hall et al., 2014) and declining grades of ores (Northey et al., 2014). Therefore, we can expect a slowdown of efficiency gains and even an increase in the emission intensity of the production of agricultural inputs (Aguilera et al., 2015), even if substantial improvements in some GHG-intensive areas such as China can still be expected (Zhang et al., 2013).

Synthetic or inorganic chemicals

Fertilizers are probably the major agricultural input in terms of energy consumption and GHG emissions. They represent about 2% of primary energy consumption in the world (Aguilera et al., 2015).

Nitrogen (N) fertilizers are the most energy consuming of macronutrient fertilizers, mainly due to the high amount of energy required to fix atmospheric N_2 (ecoinvent Centre, 2007, Kool et al., 2012), despite substantial improvements have been made in the last century (Aguilera et al., 2015). N fertilizers are also those applied in highest quantities (FAOSTAT, 2016), thus being an important source of GHG emissions. Most GHG emissions from N fertilizer manufacture are related to the use of fossil fuels, mainly natural gas for NH_3 production, but also other fossil fuels along the full production process, including fuel production, NH_3 synthesis, commercial fertilizers production, packaging, transport and retail. Moreover, the industrial production of nitrate is associated to N_2O emissions, which are added to emissions related to fossil energy consumption. Overall, the world average carbon footprint of N fertilizers may represent 5.7 kg CO_2-eq/kg N, ranging between 3.3 kg CO_2-eq/kg N for ammonium sulphate to 9.5 kg CO_2-eq/kg N for calcium ammonium nitrate (Kool et al., 2012). There also important differences between world regions. For example, average emissions from urea production are estimated to represent 3.5 kg CO_2-eq/kg N in Western Europe, versus 7.4 kg CO_2-eq in China+India and 5 kg CO_2-eq/kg N in the world (Kool et al., 2012). The carbon footprint of global average N fertilizer production was estimated at 5.7 kg CO_2-eq/kg N (Kool et al., 2012), which might be compared with the carbon footprint of direct N_2O emissions from fertilizer application to soils, representing 4.7 kg CO_2-eq/kg N if we assume the default emission factor of 1% proposed by IPCC (2006). Assuming the average carbon footprint of 5.7 kg CO_2-eq/kg N, and taking into account that global N fertilizer consumption was 108 Tg N in 2008 (FAO, 2016), global GHG emissions from N fertilizer production would be 0.6 Pg CO_2-eq in 2008. This represents 1.31% of 47 Pg CO_2-eq global GHG emissions in that year (IPCC, 2014).

Phosphate (P_2O_5) fertilizers production employs fossil energy for mining and beneficiation of phosphate ore, sulfur production (byproduct at crude oil refinery), sulfuric acid production, superphosphate manufacturing, and granulation and transport of the final product. The exothermic reaction of rock phosphate and sulfuric acid is employed as a source of energy in modern plants (Jenssen and Kongshaug, 2003). Phosphates fertilizer manufacture can follow wet or thermal routes. The main sources of environmental impacts can greatly differ between the two routes (da Silva and Kulay, 2005), depending on factors such as distance from mines to factories or the electricity generation mix. GHG emissions from average P fertilizer production ranged from 1.8 to 11.0 kg CO_2-eq/kg P_2O_5 in various studies reviewed by Linderholm et al. (2012). Transport often represents an important share of P fertilizers carbon footprint because of the low concentration of phosphate rock and because the world reserves are concentrated in few places. The world average carbon footprint of P fertilizers production has been estimated at 1.35 kg CO_2-eq/kg P_2O_5 (Kool et al., 2012). Taking into account that global P fertilizer consumption was 41 Tg P_2O_5 in 2008 (FAO, 2016), this represents 0.12% of 47 Pg CO_2-eq global GHG emissions in that year (IPCC, 2014).

Potash (K_2O) fertilizer production is mainly based on mining soluble ores of potash such as sylvite KCl), which are predominantly found in large deposits in the northern hemisphere (Ciceri et al., 2015. The fertilizer production process involves fossil energy use in mining and processing of the ores, transport and packaging of the final products. The average carbon footprint of potash fertilizers has been estimated to represent 1.23 kg CO_2-eq/kg K_2O, ranging from 0.23 to 1.91 kg CO_2-eq/kg K_2O (Kool et al., 2012). Taking into account that global K fertilizer consumption was 41 Tg K_2O in 2008 (FAO, 2016), this represents 0.08% of 47 Pg CO_2-eq global GHG emissions in that year (IPCC, 2014). Thus, total emissions from N, P and K fertilizer production would amount to 1.51% of global GHG emissions.

Lime is an important input in cropping systems with acidification problems. The global average carbon footprint of lime is 0.074 kg CO_2-eq/kg lime. **Micronutrients** are also applied to agricultural soils to replenish extractions with yields and overcome soil deficits. There is a lack of published information related to the carbon footprint of micronutrient fertilizers.

Synthetic **pesticide** production is a very energy-intensive process (Green, 1987, Bhat et al., 1994, Audsley et al., 2009), although it has been specifically studied in very few occasions, and most times they have used the inventory data of Green and McCulloch (1976) and Green (1987) study. Unlike other industrial processes, the energy efficiency of average pesticide production does not seem to have improved along history, but rather the opposite trend has been observed: old, simple pesticides are being substituted by more complex ones (Audsley et al., 2009). Despite many studies and databases (e.g. Audsley, 2003, ecoinvent Centre, 2007) classify pesticides by function (such as herbicides, insecticides and fungicides) and by chemical family (such as phenoxi, benzoic, acetamide, triazine, organophosphorus or carbamates), Audsley et al. (2009) found that this factor was not a good predictor of energy use and GHG emissions from pesticide manufacture. Aguiera et al. (2015), using the data from Audsley et al. (2009), estimated that the embodied energy of the average pesticides used in 2010 was 447 MJ/kg active ingredient, which would translate into 30.8 kg CO_2-eq/kg active ingredient using Audsley et al. (2009) energy carbon intensity factors. Zhang et al. (2011) provide a rough estimate of a global use of 4.6 Tg of chemical pesticides. Assuming this value and the above estimated emission factor of 30.8 kg

CO_2-eq/kg pesticide, this represents 0.30% of 47 Pg CO_2-eq global GHG emissions in 2008 (IPCC, 2014).

Energy

Fuel consumption in agriculture is much lower than in transport, but it is still a significant source of GHG emissions. This includes emissions from fuel combustion and fuel production due to fuel consumption by agricultural machinery, including self-propelled machinery such as tractors and combine harvesters, static engines such as water pumps, and other uses such as heat production for barnyards, greenhouses or grain drying. For example, total fuel consumption emissions of various crops of an arable crop rotation in Japan ranged between 424 and 826 kg CO_2-eq/ha yr (Koga et al., 2003). Transport of farm inputs or of organic materials within the farm can also consume significant amounts of fuel. For example, Wiens et al. (2008) estimated that liquid pig manure transport distance could only be increased to 12.3 km before the energy cost of manure N was equivalent to urea N.

Fuel **combustion** emissions are mainly due to the release of fossil carbon to the atmosphere, although trace gases are also emitted. Agricultural machinery fuel combustion emissions represent about 3.2 kg CO_2-eq/kg fuel (either diesel or gasoline) (MAGRAMA, 2013), which, taking into account the density of diesel and gasoline, is equivalent to 2.7 kg CO_2-eq/l diesel and 2.4 kg CO_2-eq/l gasoline. Direct emissions from renewable fuels such as biomass only include emissions of trace gases (N_2O and CH_4). They represented 0.14 kg CO_2-eq/kg biomass in Spanish agriculture (MAGRAMA, 2013). Fuel **production** emissions include all processes involved: extraction, transport of raw resource, processing (refining of petroleum products) and transport of the refined product. Fuel production energy requirements (and associated emissions) are now increasing due to depletion of the best resources (see Section 1.1.1). For example, the energy requirements of gasoline increased from 0.24 MJ/MJ in 1990 to 0.28 MJ/MJ in 2010 (Aguilera et al., 2015), which would approximately translate into 15.1 and 17.4 kg CO_2-eq per GJ gasoline, respectively.

Electricity is mainly used in agriculture for water pumping in irrigation and for drying crops (Cleveland, 1995), although it also has other uses such as lighting and ventilation of livestock facilities (Nacer et al., 2016) and operation of composting plants (Andersen et al., 2010). A carbon-intensive composition of the electric grid can lead to higher GHG emissions from electricity-based water pumping than from diesel-based pumping systems (Tyson et al., 2012). GHG emissions from electricity production can be the dominant source of environmental impacts in cropping systems with a heavy electricity usage and a carbon-intensive electricity grid, such as horticultural systems irrigated with groundwater in Greece (Foteinis et al., 2016). The carbon footprint of electricity production includes direct emissions from plant operation and indirect emissions from fuel provision and infrastructure (Turconi et l. 2013). The relative share of each stage and the resulting total carbon footprint have a wide variability, depending on the type of technology employed and its specific characteristics. Herein I present a brief overview of carbon footprint values for electricity production sources estimated by various meta-analyses of published studies. For thermal electricity generation with fossil fuels, the carbon footprint is typically highest for **coal,** ranging from 675 to 1,689 g CO_2-eq/kWh with a median value of 1,001 g CO_2-eq/kWh (Whitaker et al., 2012). Carbon capture and storage can reduce these emission levels but they also increase many other environmental impacts (Schreiber et al., 2012). Coal is followed by **diesel**, with 720 g CO_2-eq/kWh (Amponsah et al., 2014). Emissions from **natural gas**-based electricity are typically lower, although there are significant differences between natural gas-fired combustion turbines, with a median value of 670 g CO_2-eq/kWh, and combined-cycle systems, with a median value of 450 g CO_2-eq/kWh (O'Donoughue et al., 2014). Gas leakages leading to fugitive CH_4 emissions could drastically increase the GWP of natural gas-based electricity, negating climate benefits of natural gas over coal if they represent more than 5.2-9.9% of natural gas production (Hausfather et al., 2015). Average fugitive CH_4 emission rates from the natural gas industry have been estimated to be 2-4% (Schwietzke et al., 2014). GHG emissions from **nuclear**-based electricity production are the lowest among non-renewable electricity sources, ranging between 10 and 131 g CO_2-eq/kWh, with an average of 65 g CO_2-eq/kWh (Lenzen, 2008), although another quantitative review offered a lower median value of 12 g CO_2-eq/kWh (Warner and Heath, 2012). The estimations of the carbon footprint of **renewable** electricity sources typically lead to lower values than those of fossil fuels. Asdrubali et al. (2015) estimated average emissions of 9, 12, 29, 31 and 34 g CO_2-eq/kWh for wind, hydropower, photovoltaic, concentrated solar power and geothermal electricity production, respectively. Worst-case scenario emission values for renewable technologies might rise up to 124, 75, 300, 150 and 78 g CO_2-eq/kWh for wind, hydropower, photovoltaic, concentrated solar power and geothermal electricity production, respectively (Amponsah et al., 2014), still significantly lower than fossil fuels. Biogenic methane emissions from reservoirs have been overlooked in many studies, but they could significantly increase the carbon footprint of **hydropower**. The global average emissions of biogenic methane have been estimated to represent from 3 g CH_4/kWh (Hertwich, 2013) to 5.7 g CH_4/kWh (Li et al., 2014). Furthermore, hydroelectricity emissions could be even larger if decommissioning emissions are also accounted for (Pacca, 2007). Additional energy costs are added for transporting electricity, including infrastructure production and maintenance and grid losses (Aguilera et al., 2015).

Capital goods

Emissions from the production of capital goods mainly include machinery manufacture, construction of irrigation infrastructure, greenhouses and buildings. These emissions are produced off-farm and include the production of raw materials and fuels, manufacture of the components and construction or manufacture of the final good. Their embodied energy and associated GHG emissions would depend on the energy efficiency and carbon intensity of all stages of the production process and also of the useful life of the capital good. The latter factor has a large impact on the resulting annual embodied energy of capital goods such as irrigation infrastructure (Diotto et al., 2014).

Machinery production emissions usually represent a small fraction of the total GHG balance of agricultural systems (West and Marland, 2002), despite it may be a very significant input from an economic point of view (Mobtaker et al., 2010).

Irrigation infrastructure includes both upstream infrastructure such as dams, channels and pipes, and on-farm infrastructure such as ditches, pipes, and emitters. Upstream irrigation GHG emissions have not been characterized as far as I know. High GHG emissions from hydroelectric dams due to biogenic CH_4 emissions (see above in this section), however, should warn us about potentially large emissions from dam irrigation water. Lal (2004b) estimated the carbon footprint of the on-farm installation of different irrigation systems employing the inventory data from Batty and Keller (1980). The values ranged from 34 kg CO_2-eq ha/yr for surface irrigation systems without runoff return system, 90 kg CO_2-eq/ha/yr if they had runoff return system to 311 kg CO_2-eq/ha/yr for drip irrigation systems. The carbon footprint of sprinkler systems can vary widely from 60 kg CO_2-eq/ha/yr for hand-moved sprinkle to 445 kg CO_2-eq/ha/yr for solid set sprinkle, with intermediate values for other systems such as traveler, center-pivot or permanent sprinkle. These values can be lower or higher than those associated to energy use depending on the energy intensity of water (Sanz-Cobena et al., Under Review), which depends on the type of system and lift height (with sprinkler requiring more energy to pressurize, and furrow systems requiring more energy to lift due to higher water consumption).

Greenhouses and other usages of plastic for crop protection, such as tunnels and mulches, can be a major source of GHG emissions in intensive crop production systems (Girgenti et al., 2013, Romero-Gámez et al., 2014). Glass greenhouses are much more energy and carbon intensive than plastic greenhouses, with average values of 2.9 kg CO_2-eq/m^2, versus 0.5-1.3 kg CO_2-eq/m^2 for multi-tunnels and parral-type greenhouses, and 0.4-0.6 kg CO_2-eq/m^2 for low-tunnels (Anton et al., 2014).

Organic inputs

The emissions associated to the production of **organic inputs** are usually accounted as residue management emissions, given that the residue would have to be managed in any case. Transport can also imply substantial energy costs for organic fertilizers (see Fuels emissions subsection). The main types of organic residues include manure (both liquid and solid), agro-industry waste (such as olive mill waste and winery waste), municipal solid waste and sewage sludge. The management of organic residues is responsible for the emission of large amounts of trace GHG gases, mainly CH_4 and also N_2O.

GHG emissions from **residue management** strategies should be assessed taking into account GHG emissions from management but also the destiny of the products of the processed residue and their implications for GHG emissions. For example, if the residues are used for **animal feeding**, it is necessary to consider what products does it substitute and what is the effect on animal productivity and biogenic GHG emissions (Pardo et al., 2016); for **energy production**, what is its emission intensity, how much nutrient is recovered and what is the emission intensity of the energy it substitutes (Aguirre-Villegas et al., 2014); for **soil application**, how stabilized is its carbon (Thomsen et al., 2013) and what is the effect on N_2O emissions.

1.3 Greenhouse gas mitigation in agriculture

1.3.1 Classifying agricultural mitigation practices

The role of agriculture in anthropogenic GHG emissions was discussed in previous sections. Its relevance indicates that agriculture should also hold a central role in climate change mitigation strategies. In a complex, inter-linked world, these strategies should be able not only to address the mitigation component of agricultural sustainability, but also the possible trade-offs, co-benefits and barriers (Bustamante et al., 2014, Paustian et al., 1998, 2016, Sanz-Cobena et al., Under Review).

In this section I overview agricultural GHG mitigation practices, with a focus on crop production. Given the high number and diversity of practices, it is useful to classify them according to different criteria. Garnett (2014) classify sustainable food security measures in three perspectives: efficiency, demand constraint and food system transformation. This classification is oriented towards the food system as a whole. In this dissertation the focus is on agricultural production, but also considering other measures and processes with impacts on agriculture, and with a special emphasis on organic farming practices. Thus, agricultural GHG in Table 1.1 are classified according to three criteria: i) **technological, management or agro-food system/structural practices**, based on the degree of the technological and novelty components of the practice. This criterion is somewhat arbitrary, as all measures have a technological component and are subject to technology-based improvements, but it can still be illustrative; ii) **target system of the measure**, including field, animal (which are summarized in the table but not discussed in the text) or the whole farm; iii) **compatibility with organic farming**, depending whether they can comply or not with organic farming certification rules, chiefly non-use of synthetic chemicals and genetically modified organisms. In the case of agro-food system/structural measures, we have also considered compatibility with organic farming principles and recommendations (i.e. local consumption); Table 1.1 also includes measures at the agri-food system level and structural measures based on demand changes, with implications for agricultural GHG mitigation, which will be briefly discussed in Section 5.3. On the other hand, despite some livestock mitigation measures are mentioned in Table 1.1, they won't be further discussed for space reasons.

Table 1.1 Greenhouse gas mitigation practices in agriculture and the food system

Technological/management/ Agro-food system-structural	**Target system**	**Organic-compatible**	**Organic-non compatible**
Technological (employ novel or not widespread technologies)	Field	Precision farming (through monitoring and Information and Communication Technologies) Irrigation Fertilization Tillage Allelopathy Promote roots Biochar	Inhibitors Genetically Modified Organisms
	Animal	Natural dietary additives	Chemical dietary additives Genetically Modified Organisms
	Farm	Farm waste (crop residues, manure) resource recovery (feed, energy, nutrients, carbon…) Biogas and biofertilizer Pyrolysis: energy and biochar Renewable energy production and use Electricity Biofuels for self-consumption	Chemicals applied to slurry storage
Management (employ common technologies,	Field	Reduced/no tillage without herbicides	Reduced/no tillage with herbicides

sometimes used in a different way)		Organic N fertilizer application method, timing and placement Improved rotations Longer rotations More legumes Cover crops/Less bare fallow Intercropping/Agroforestry Erosion reduction measures Restoration of degraded land Improve yields	Synthetic N fertilizer application method, timing and placement Chemical-based methods of yield improvements
	Animal	Pasture management Animal diet management Increase animal productivity	
	Farm	Enhanced multi-functionality Reconnection of livestock and cropland Farm waste (crop residues, manure) resource recovery (feed, energy, nutrients, carbon…) Stubble grazing Retention Traditional biomass use Compost Other traditional renewable energy (human, animal, mechanical wind, passive solar)	
Agri-food system/structural	Technical	Agri-food system waste (agro-industry, urban solid and wastewater) resource recovery (feed, energy, nutrients, carbon, biomaterials)	
	Demand	Local consumption (reduced transport) Reduced waste Dieatry changes (reduced animal component) Global trade (cultivation in best suited areas to optimize yields)	

1.3.2 Technological measures

Urease and nitrification inhibitors can increase yields and N use efficiency of crops, although their effect is variable (Abalos et al., 2014). Meta-analysis studies indicate that nitrification inhibitors reduce N_2O emissions, but the effect of urease inhibitors is less clear (Akiyama et al., 2010, Abalos et al., 2016). Urease and nitrification inhibitors have shown the potential to reduce both N_2O emission factors and yield-scaled emissions in Mediterranean cropping systems (Sanz-Cobena et al., 2012, Cayuela et al., Under review). Their effects on other components of the GHG balance and further environmental effects, however, are much less known.

Genetically modified organisms (GMO) potentially usable for GHG mitigation in agriculture include a wide spectrum of traits, from transgenic plants expressing nitrous oxide reductase (Wan et al., 2014), to drought resistant plants reducing water demand. In spite of this, the use of GMO in the world has mostly focused on just herbicide resistant and insecticide producing plants. In particular, the expansion of glyphosate-resistant plants has led to a huge expansion of glyphosate use, up to representing most herbicide use in crops such as soybeans in the US (Bonny, 2008). Herbicide resistant plants would facilitate the application of no tillage practices. These practices, however, are being questioned as an effective GHG mitigation strategy (Sections 1.2.2, 1.2.3 and 1.3.3). On the other hand, the production of their seeds is highly intensive in technology, which involves a high use of energy and resources (Rotolo et al., 2015), potentially affecting the GHG emission balance. GMO can also threaten cultivated biodiversity (Pineyro-Nelson et al., 2009), and thus climate change adaptation potential.

Under the term **precision farming** I refer to practices aimed to increase the efficiency of resource use in the farm through various technological means. Most of these practices can be applied in organic farming systems. GHG mitigation with these practices would be achieved through reduced emissions due to the production of inputs, although sometimes field emissions are also affected. They usually involve the use of information and communication technologies. This means that their use also involves energy and resource consumption for the manufacture and operation of the communication systems, which should be considered in environmental assessments (Park ad Malakon, 2013). We can identify three groups of precision farming practices, according to the resource they address:

Water, through **improved irrigation** systems. The water-energy nexus, and its impacts on GHG emissions, is clearly illustrated by the efforts in improving irrigation systems. Increased water efficiency could lower energy consumption because less water has to be pumped, and the decline of groundwater level might be avoided (Karimi et al., 2012). However, the substantial improvement in water use efficiency achieved with the modernization of irrigation systems (substituting surface irrigation with sprinkler and drip systems) was made through the conversion of open-channel systems to pressurized networks, which are associated to higher energy costs

(Fernandez-Garcia et al., 2016). This way, the modernization of irrigation in the Mediterranean region has been estimated to potentially save 8 km^3 of water, but CO_2 emissions associated to irrigation energy would rise by 135% (Daccache et al., 2014).

Nutrients, through **improving nutrient use efficiency**. Adjusting nutrient doses to crop needs (e.g. Arregui and Quemada, 2008) through monitoring of soil mineral nutrients content, split applications of fertilizers, use of controlled-release fertilizers or other practices, is one of the most effective strategies to reduce N_2O emissions, which could also have benefits for the mitigation of indirect N_2O and emissions from the production of fertilizers, due to lower fertilizer use.

Fuel and other inputs, through **guidance systems** reducing overlapping of field tasks. GPS-based guidance systems can avoid waste of fuel, fertilizer, pesticides and seeds (Zier et al., 2008, Kroulik et al., 2011).

Other novel strategies for GHG mitigation compatible with organic farming are the use of **allelopathy** to control weeds, either by using natural allelopathic plant interactions or by using allelochemicals as natural herbicides (Singh et al., 2003, Narwal, 2010, Arantini et al., 2012), which may facilitate tillage reduction and yield improvement in organic systems; and **promoting root growth** through management practices and selection of crop **cultivars with large, deep root** systems. Despite roots are not usually considered subject to management, recent assessments have identified breeding of crops with larger and deeper root systems as a very promising way to mitigate agricultural GHG through the increase of soil C inputs and through yield improvements in poor environments (Lynch, 2007, Lynch and Wojciechowski, 2015, Paustian et al., 2016). **Biochar**, the solid product of the incomplete combustion of biomass, is another promising strategy to reduce GHG emissions. Despite archeological evidence indicates that it has been used millennia ago in Central Amazonia ("Terra Preta de Indio"), it can be considered as a "technological measure", given that its current use is very incipient, and its modern production usually involves modern technology and knowledge. Biochar potentially contributes to the soil C sink (Woolf et al., 2010, McHenry, 2009), yield increases, although very variable depending on specific conditions (Vaccari et al., 2011, Jeffery et al., 2011), and reduction of N_2O emissions (Cayuela et al., 2014). It is necessary to take into account the life cycle GHG emissions of biochar production systems: from this perspective, biochar may only be an effective GHG mitigation strategy if it is implemented as distributed systems (reduced transport impacts) using waste biomass (reduced feedstock impacts) (Robert et al. 2010).

At the **farm level**, there is a large potential for GHG mitigation through **residue management** (mainly manure), involving an important technological component. Some alternative residue management methods are liquid storage, solid passive storage (including manure stockpiling and MSW landfilling), drying or composting. Some technologies achieve multiple uses of the residue, such as **biogas** production, which generates "digestates", conserving all the initial nutrients, that can be applied as fertilizers (Meyer-Aurich et al., 2012, Vaneeckhaute et al., 2013), **pyrolysis** to obtain gaseous and liquid fuels and biochar which can be applied to the soil (Zabaniotou et al., 2000).

Production of **renewable energy**, chiefly for self-consumption purposes, could be a way to reduce fossil energy use and associated emissions and to achieve a greater self-sufficiency, given their lower GHG emission intensity (Section 1.2.4) and their potential for implementation in most places. Bio-based fuels such as ethanol, methanol, biodiesel, raw oil, biogas or hydrogen have a large GHG reduction potential when used for energy self-sufficiency, particularly when they are made of residues rather than dedicated crops, although a number of technical and socioeconomic barriers should be overcome (Bernesson et al., 2006, Halberg et al., 2008, Ahlgren et al., 2009, Smyth et al., 2010, Pugesgaard et al., 2014). A special attention has to be paid to the land cost of self-production of the fuel, which may indirectly cause LULUCF emissions.

Some approaches **sequentially employ different technologies**, potentially increasing overall efficiency, such as solar-enhanced efficiency of biogas production (Dong et al., 2013), biogas production and pyrolysis of solid digestates (Schouten et al., 2012) or even more integrated approaches encompassed in the term "biorefinery" (Tonini et al., 2013).

1.3.3 Management measures

The main aim of **conservation tillage**, including **no-tillage** and **reduced tillage,** is to avoid harmful impacts of tillage on soil quality and protect it against erosion. However, tillage effects on soil GHG emissions are not straightforward, and they can lead to a reduction or an increase in emissions depending on the specific situation. Tillage effects on N_2O were discussed in Section 1.2.2 and effects on C sequestration in Section 1.2.3. This practice could also contribute to reduce fuel consumption and associated GHG emissions (Robertson e al. 2000, Guardia et al., 2016). The major alternative to tillage in cropping systems is **weed control with herbicides**. This has been facilitated by GMO crops designed to resist herbicides (Section 1.3.2). The management challenges of conservation tillage under organic farming are much greater due to the avoidance of herbicides, but a wide range of management practices can help to tackle them (Peigne et al., 2007, Altieri et al., 2011).

Diversification allows a more efficient use of the agroecosystem resources due to complementarities achieved among different crops. Diversification practices based on the alternation of different crops are called rotations. **Improved rotations** usually include many crop types and reduce bare fallow through the use of cover crops or by intensifying the rotation (producing an additional cash crop). **Cover crops** refer to the cultivation of crops whose biomass is not extracted from the system. In arable systems, cover crops are usually substituting bare fallows, either in the winter period between two summer crops, or in certain years of rotations of winter crops. In woody systems, cover crops may be cultivated between trees or covering the entire soil surface. Cover crops have been shown to be an effective practice to increase SOC levels in agricultural soils (Poeplau and Don, 2015). Their effect on N_2O emissions is not so clear, however. In a recent meta-analysis, Basche et al. (2014) found lower N_2O emissions with cover crops in 45% of the studies and higher emissions in 60% of the studies, although the net effect was close to zero in studies measuring for a full year. **Longer rotations**, with a higher number of crops and more years between the cultivation of the same crop, can improve yields while decreasing needs for fertilizers and pesticides (Benneth et al., 2012, Davis et al., 2012). The benefits are higher if they include legumes (Albizua et al., 2015). **Legumes** have clear benefits in agro-ecosystems due to their ability to fix atmospheric N_2, thus decreasing the need for external fertilizers and improving the productivity of other crops (Peoples et al., 2009: Anglade et al 2015)). The fertility benefits of legumes do not end with biological N fixation: they have also been shown to promote root growth in the following crop (Gaiser et al., 2012). Furthermore, legume cultivation is associated to lower N_2O emissions than that of fertilizer crops. Overall, these benefits can lead to very significant GHG mitigation through the cultivation of legumes (Jensen et al., 2012).

Other methods of crop diversification are based on **intercropping**, or the simultaneous cultivation of various crops, and it is usually called agro-forestry when it includes woody species. Intercropping can have multiple benefits with implications for GHG emissions mitigation. One benefit is that the roots systems and nutrient uptake might be enhanced, triggered by competition between species and complementarity of root systems. For example, walnuts roots grew deeper in the presence of herbaceous winter crops (Cardinael et al., 2015), and intercropped maize/bean/squash systems greatly increased N uptake and biomass production (Postma et al., 2012). Intercropping can also be used to exploit allelopathic interaction between plants, improving their agronomic performance (Section 1.3.2). Another benefit, which is derived from the others, is an improvement of yields, which is favored by the presence of legumes and by a careful combination of species (Pappa et al., 2012). **Agroforestry** is a particularly interesting strategy for GHG mitigation (Paustian et al., 2016), due to the accumulation of carbon in the living biomass and potentially also in the soil (Albrecht and Kandji, 2003, Nair et al., 2009). A recent meta-analysis indicates that agro-forestry is associated to an average sequestration of 7.2 Mg C/ha yr, 70% of which is in standing biomass and 30% in the soil, while no differences in soil N_2O emissions and CH_4 oxidation were found (Kim et al., 2016). Agroforestry also greatly improves the multi-functionality of the system (Mbow et al., 2014): production diversifies, including woody biomass that can substitute fossil fuels, and other products, while the provision of ecosystem services is enhanced, such as biodiversity conservation and climate change adaptation (Rahn et al., 2014).

Many **residue management** measures employ very simple or consolidated technology. Residue grazing, incorporation of residues to soil, traditional biomass use, traditional use of other renewable resources. Residue grazing can make use of the feed value of residues while avoiding harvesting and manure handling costs. The net effects of **grazing** on SOC are more positive than baling (Banco-Canqui et al., 2016). In dryland systems with appropriate management, stubble grazing can maintain or even increase SOC levels even when compared to stubble retention (Barsotti et al., 2013), and without negative effects on yields (Lessen et al., 2013), although the effects of variations in the composition of animal diet on enteric CH_4 and manure management emissions should also be considered. On the other hand, residue **retention** has clear advantages for the protection against erosion, for SOC storage and for nutrient cycling, potentially improving fertilizer nutrient use efficiency (Murphy et al., 2016). Biomass for **thermal energy** purposes has traditionally represented a key use of woody crop residues when they were available, with a special significance for the Mediterranean region (Infante-Amate et al., 2014). This logic can be followed by modern approaches, although, as in the other cases, the alternative destinies and their effects on agroecosystem properties have to be taken into account. Last, **composting**, the aerobic fermentation of organic residues, is another typical management practice of organic cropping systems (Goh, 2011, Leu, 2014). Compost is a type of organic amendment associated to particularly high C sequestration rates and fertility improvements (Diacono and Montemurro, 2010). This is probably due to a greater stabilization of carbon in composted materials, which is in line with studies finding higher humification rates for compost than for other organic materials (Katterer et al., 2014), although this recalcitrance could negatively affect crop yield (Katterer et al., 2014). On the other hand, a recent meta-analysis has shown that the composting process is less emission-intensive than other organic waste management practices (Pardo et al., 2014), which would further support the use of composts to mitigate GHG. Less information is available on composting effects on soil N_2O emissions (Favoino and Hogg, 2008).

Yield increase is often proposed as a major GHG mitigation option (e.g. Snyder et al., 2009, Burney et al., 2010, Katterer et al., 2012, Valin et al., 2013, Godfray and Garnett, 2014, Lamb et al., 2016, Paustian et al., 2016), stressing the importance of avoiding emissions from indirect land use changes driven by agricultural expansion. Yield increase is not a management practice itself, but the outcome of practices related to management intensification. These practices range widely from chemical-based measures to other based on organic inputs or other based on the intensification of internal loops, with very varied impacts on agricultural GHG emissions, and many studies call for this intensification to be sustainable (Valin et al., 2013, Godfray and Garnett, 2014). Moreover, increasing productivity may have trade-offs, such as those arising from feeding high-quality feedstock to improve productivity of animal production, which leads to competition with human food production (Davis et al., 2015). On the other hand, as stressed by

Bustamante et al. (2014) and Hertel et al. (2014), yields improvements in the real world could foster feedbacks leading to "rebound effects" such as higher meat consumption, potentially negating land sparing effects of higher yields, at least in certain situations. Indeed, demand-based approaches to sustainable food security are an alternative approach to avoid LULUCF emissions and lower agricultural GHG (Section 5.3.2). The combination of these demand-side measures with strategies such as feeding less food-competing feedstock to livestock have been found to be a good complement to strategies focused on increased production efficiency (Schader et al., 2015).

1.3.4 Organic farming

The International Federation of Organic Agriculture Movements (IFOAM) defined organic farming as "Organic Agriculture is a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic Agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved." (IFOAM, 2005). The use of synthetic chemicals such as fertilizers and pesticides is specifically excluded from organic farming systems, which thus rely on symbiotic fixation and organic matter management as source of fertility. Although the variability is high, organic farming has been linked to generally higher biodiversity levels (Birkhofer et al., 2008, Tuck et al., 2014), reduced soil erosion (Reganold et al., 1987), increased energy efficiency (Gomiero et al., 2008, Smith et al., 2015), improved soil quality (Gomiero et al., 2011), better quality of produced food (Zalecka et al., 2014) and better economic performance (Crowder et al., 2015) than conventional farming, being generally lower yields its major drawback (Seufert et al., 2012, de Ponti et al., 2012, Ponisio et al., 2014). Overall, organic farming has been shown to deliver social and environmental benefits despite lower yields (Reganold and Watcher, 2016). In general, organic practices could be considered "preventive" measures, as opposed to "curative" measures addressing the consequences of environmental impacts (Garnier e al. 2014).

Scialabba and Muller-Lindenlauf (2010) identified 3 major ways in which organic agricultural systems may contribute to **climate change mitigation**: reduction in N_2O emissions through careful management of nutrients, abstention from the use of mineral fertilizers and the associated emissions related to their production, and enhanced carbon sequestration.

In a global meta-analysis, Skinner et al. (2014) found that organic farming indeed led to significant reductions in area-scaled **N_2O emissions**. However, due to lower yields, it increased yield-scaled N_2O emissions. A yield gap lower than 17% was required to equalize those means. This shows that yield reductions could compromise the environmental benefits of management changes such as those applied with organic farming. On the other hand, organic cropping systems were also related to higher methane uptake than conventional cropping systems (Skinner et al., 2014).

In another global meta-analysis, Gattinger et al. (2012) reported higher SOC levels under organic farming, averaging 0.18% higher **SOC concentration**, 3.5 Mg C higher SOC stocks and 0.45 Mg C higher SOC sequestration rates. The authors observed positive effects of organic farming practices on SOC even restricting the analysis to systems without external organic inputs, which contradicts the idea that carbon sequestration benefits of organic farming could be due to a disproportionate use of external organic inputs (Leifeld and Fuhrer, 2010, Leifeld et al., 2013). However, as was also acknowledged by Gattinger et al. (2012, 2013), enhanced carbon stocks do not necessarily imply climate change mitigation (see Section 1.2.3).

As described in Section 1.2.4, synthetic fertilizers and pesticides require a lot of **energy** for their production processes, making them also very carbon intensive. Thus, their avoidance in organic farming systems can lead to GHG mitigation, although the effects of changes in the use of other inputs (such as machinery and fuel) and in yields also have to be considered to determine their mitigation potential. Meta-analytic studies show that, despite there are also some exceptions, the energy efficiency is consistently increased in organic systems (Gomiero et al., 2011, Lynch et al., 2011, Smith et al., 2015), and the reliance on renewable energy sources is also higher in organic farming systems (Smith et al., 2015).

Thus, the overall effect of organic farming practices on the GWP of cropping systems is typically defined by lower **emission intensity on an area basis**, due to the avoidance of chemical inputs and lower N inputs, but their performance is more variable **on a product basis**, partly because lower yields may offset the area-based reductions (Lynch et al., 2011, Gomiero et al., 2011, Meier et al., 2015). Further implications of lower yields are discussed in Section 1.3.3, while possible ways to solve this problem in organic systems are discussed in Section 5.2.1 and through demand-based measures in Section 5.3.2.

1.4 The Mediterranean context

1.4.1 Climate and territories

The **Mediterranean climate** is characterized by dry, hot summers and wet, mild winters. Therefore, an important characteristic of this climate is seasonal dryness, and many of its subtypes are classified as semi-arid. An accepted definition states that at least 65% of annual rainfall occurs in winter and the annual precipitation ranges from 275 to 900 mm, with an average winter temperature is below 15ºC, but the number of hours per year with temperatures below 0ºC does not exceed 3% of the total (Aschmann, 1973).

Territories with Mediterranean climate are generally located on the west coasts of continents and between latitudes 30º and 45º north and south of the equator, with over half of its area found in the Mediterranean Sea Basin and the rest distributed in four other different regions of the world, namely California (USA), Central Chile, the Cape region of South Africa, and South-West Australia (Aschmann, 1973). The range of the Mediterranean climate strictly following the definition by Aschmann (1973) has been estimated at 1.5 million km² (Klausmeyer and Shaw, 2009), but a more accepted map of the Mediterranean biome covers 3.2 million km² (Olson and Dinerstein, 2002, Klausmeyer and Shaw, 2009, Figure 1.8).

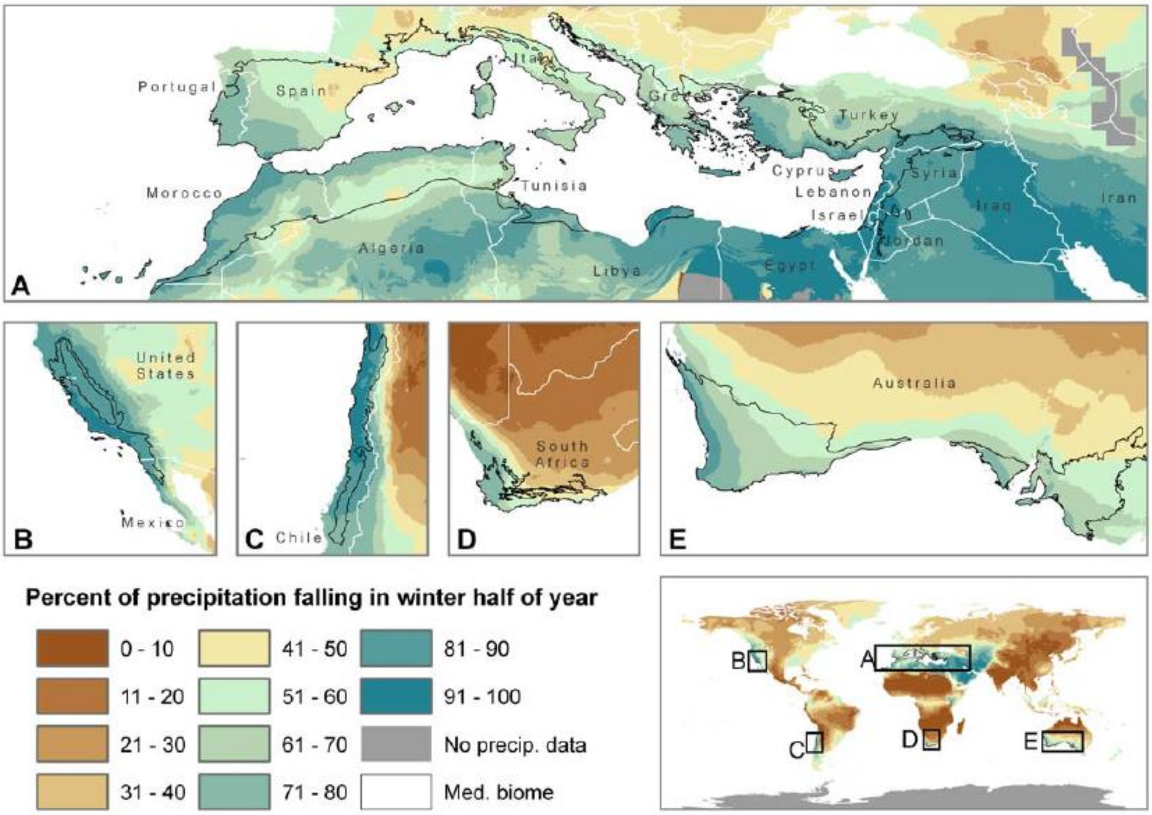

Figure 1.8 Global distribution of Mediterranean biome and percent of precipitation falling in winter half of the year. Source: Klausmeyer and Shaw (2009), elaborated with precipitation data from WorldClim dataset and Mediterranean biome distribution from Global 200 database (Olson and Dinerstein, 2002)

The **soils** in Mediterranean climate areas typically have low organic matter content due to limited precipitation, associated low organic inputs, and relatively high temperatures. In an analysis of the SOC content of global ecoregions, Stockmann et al. (2015) found that Mediterranean forests and scrublands were the biome with the second lowest average SOC content in the topsoil (0-10 cm), only after deserts. Topsoil SOC content was about 2% in the Mediterranean biome, while the world average was at ca. 3.8% C.

1.4.2 Mediterranean ecosystems and agroecosystems

The temporal gap between maximum irradiance and temperature (early summer) and maximum water availability (winter) is responsible for the typically low **productivity** of Mediterranean natural ecosystems and rainfed crops. In spite of this low productivity, however, the Mediterranean biome is a **biodiversity** hotspot, containing ca. 20% of global vascular plants in 5% of terrestrial surface (Underwood et al., 2009). Typical Mediterranean patterns have been described in many areas of ecological research, including plant physiology (González-Fernández et al., 2010), litter decomposition dynamics (Incerti et al. 2011), N biogeochemistry (Breiner et al., 2007, Lassaletta et al., 2012), limnology (Álvarez-Cobelas et al., 2005) and biodiversity (Barriga et al., 2010).

Agroecosystem diversity is also typically very high, and includes many crops which are almost exclusive of this biome, such as olive, fig and almond trees. For example, olives have been used as indicators of the Mediterranean climate in paleoclimatic studies (Moriondo et al., 2008). There are also endemic cultivars with adaptive features such as drought resistance (Galmes et al., 2011). Eichhorn et al. (2006) studied agroforestry systems of Europe, finding a higher cultivated diversity in Mediterranean areas than in temperate ones. They also found that systems from northern Europe were mainly limited by light, while the Mediterranean ones were limited by water.

Despite its high biodiversity, the Mediterranean biome is severely threatened by **human impacts**, including increasing population, urban areas and agriculture (Underwood et al., 2009). In the Mediterranean region, the per capita **ecological footprint** of the population (3.1 ha) is higher than the global average, while the biocapacity is lower (1.3 ha). This implies that the ecological deficit is much more severe than for the world as whole, averaging 1.8 ha/cap in the Mediterranean region and 0.9 ha/cap in the world (Wackernagel and Ravenel, 2012). For example, in a typical Mediterranean country such as Spain, **livestock** production is largely decoupled from its territory (Soto et al., 2016), with as much protein imported as feed as all the protein produced by its agriculture (Lassaletta et al., 2014b), which implies that a large fraction of its consumption-related GHG emissions, such as N_2O, are produced outside its territory (Lassaletta et al., 2014c).

The **intensification of agriculture** in Mediterranean areas is associated to serious environmental impacts such as erosion (Guerra and Pinto-Correia, 2016), reduced water quality (Zalidis et al., 2002), nitrogen deposition (Ochoa-Hueso et al., 2011) or biodiversity loss (Cruz et al., 2015, Goncalvez et al., 2012). In spite of this, high biodiversity levels of Mediterranean ecosystems are often maintained by human agroecosystem management. Hence, the intensification of some areas goes in parallel with the **abandonment** of traditionally managed Mediterranean agroecosystems in other areas, and the resulting forest expansion has been associated to biodiversity decreases (Bugalho et al., 2011, Marull et al., 2014, Otero et al., 2015), challenging the ideas about the restorative character of the forest transition. This decrease in biodiversity is related to the disappearance of peasant land-use mosaics and specific traditional agroecosystem arrangements such as **dehesas** (oak savannas). These diverse mosaics are the result of the coevolution of ecosystem components and human societies over several millennia (Blondel et al., 2006), and usually show higher biodiversity and carbon levels than closed forests (Verdu et al., 2000, Pinho et al., 2012, Goncalves et al., 2012) or open grasslands (Rossetti et al., 2015).

On the other hand, high solar radiation and long warm periods facilitate the achievement of high yields and make possible the cultivation of a wide range of crops under irrigated agriculture. For that reason**, irrigation** is a very important management practice in Mediterranean areas, despite limited irrigation water resources and higher water requirements than in temperate areas (Wriedt et al., 2009), and it is projected to grow still more in future (Neumann et al., 2011). Some of the impacts of irrigation are a high energy consumption and associated emissions (Daccache et al., 2014), increased risk of soil degradation through salinization (Acosta et al., 2011) and changes in the seasonality (Lorenzo-Lacruz et al., 2012) and nitrogen loads (Lassaletta et al., 2012) of rivers due to dam regulation.

1.4.3 Climate projections

The Mediterranean region is one of the most threatened by global warming, with high probabilities for the expansion of desertification processes (Safriel, 2009). Climate models have forecast a rise in temperature and a fall in precipitation in this region, with expected negative effects on water resources (Garcia-Ruiz et al., 2011, Rosenzweig and Tubiello, 1996). The rise in temperature is forecast to be higher than the global average (Giannakopoulos et al., 2009). These climate change trends are already taking place in Mediterranean areas. In Mediterranean Europe, the frequency and intensity of aggressive rainfall events has increased driven by climate change, increasing the risk of desertification and land degradation (Diodato et al., 2011). In the Iberian Peninsula, in the western Mediterranean region, drought severity has increased in the last five decades as a consequence of higher evaporative demand driven by temperature increase (Vicente-Serrano et al., 2014).

Climate change scenarios project a more adverse effect on **crop yields** and a greater risk of yield losses in Mediterranean areas than in temperate ones, due to drier and hotter conditions (Ferrara et al., 2010, Moriondo et al., 2010, Bindi and Olesen, 2011). Moreover, Mediterranean soils are threatened by declines in organic matter and degradation. The impacts of climate change, however, are complex and the uncertainties would vary widely by crop and location (Iglesias et al., 2010). For example, the effect of precipitation changes on the SOC content of Mediterranean grasslands seem to be more related to the changes in their temporal distribution than in their total quantity (Chou et al., 2008). Stronger impacts are expected in southern areas of the Mediterranean region, which are lower contributors to climate change, stressing the ethical dimension of climate change adaptation strategies (Grasso et al., 2012). Likewise, Ponti et al. (2014) found a higher vulnerability of smallholders of marginal areas to increases in olive fruit fly (Bactrocera oleae) infestations driven by climate change in the Mediterranean basin, although there was a high spatial variability in infestation trends. Climate change will impact both rainfed and irrigated agriculture. It will impact rainfed agriculture by decreasing water availability and increasing heat stress, counterbalancing the benefits on photosynthesis associated to increased atmospheric CO_2 concentrations (Chartzoulakis and Psarras, 2005). The impacts of the decrease of water resources on irrigated agriculture will be particularly severe considering that it accounts for more than half of the value of food consumed and exported in the Mediterranean region, despite the irrigated area is only a small fraction of the total agricultural area (Iglesias et al., 2011). The Iberian Peninsula is particularly vulnerable to climate change impacts on irrigation water availability within Southern Europe (Garrote et al., 2015).

1.5 Justification: The need for an integrated assessment of GHG mitigation measures in Mediterranean cropping systems

In this introduction I have outlined the main challenges faced by global agriculture in the coming decades (Section 1.1.1). As we have seen, the world is reaching its limits in terms of resource consumption and the ability of major planetary systems maintaining Earth homeostasis to cope with human-induced alterations. In particular, agriculture severely impacts regional and global systems required for its functioning, and it is affected by the changes that are taking place. Thus, the combination of resource depletion and global change, in a context of increasing population, results in a formidable challenge requiring drastic changes in the way that food is produced. This way, the expected effects climate change, increasing climate variability and leading to more adverse conditions in many environments, particularly in very vulnerable regions of low latitudes, call for a major adaptation effort in order to alleviate the impacts. This adaptation should also encompass the challenges associated to resource depletion (Section 1.1.2).

Among the outlined challenges, given its current importance for agricultural sustainability, we have focused on the role of agriculture in anthropogenic climate change (Section 1.2). The prominence of this role points at the need to realize the mitigation potential of this activity if humanity is to meet climate change mitigation goals and avoid the onset of non-reversible warming. Agricultural GHG mitigation involves a wide range of technological and management measures (Section 1.3). These choices also have implications for the other problems associated with reaching planetary limits, particularly the need for adaptation to climate change and resource depletion, which should be take into account when assessing their potential. In this sense, the holistic approach to agricultural sustainability of organic farming and agroecology suggest that their associated practices could contribute to GHG mitigation while also meeting other socio-environmental goals. Poorer yield performance and other associated problems, however, may offset the benefits of organic practices in many situations. Therefore, organic practices and other mitigation measures should be studied considering the specific agro-climatic and socio-environmental conditions of each study area.

The Mediterranean biome (Section 1.4) is unevenly distributed around the world, but the ecosystems of this biodiversity hotspot share many distinct processes resulting from convergent adaptation. This biome is one of the most vulnerable regions to climate change, and a major agricultural production area with a potentially high contribution to GHG emissions. Specific agro-climatic conditions in this biome suggest that biochemical processes responsible for soil GHG emissions could also show a distinct pattern in Mediterranean agro-ecosystems potentially affecting the estimation of the net C footprint of agricultural products. For example, a study under Mediterranean conditions in Australia (Biswas et al., 2008) clearly point in this direction: the total C footprint of wheat decreased from 487 to 304 kg CO_2e per Mg of wheat, and the share of N_2O emissions in the total C footprint of wheat decreased from 36% to 9%, when switching the factor from IPCC to a local one. A LCA study incorporating C sequestration estimated through a gross IPCC approach suggests that the relevance of this process may be very high in the C footprint of Mediterranean crop products (Venkat, 2012), a finding that is supported by another study directly measuring C sequestration and soil GHG emissions (Guardia et al., 2016). Unfortunately, this type of integrative studies is very scarce in the literature.

The study of Mediterranean agroecosystems is also interesting in a context of increasing temperature and decreasing precipitation in many temperate areas (Trnka et al., 2010), which could lead to a process of "mediterraneization". This idea is supported by modeling studies projecting that the Mediterranean climate range could expand by 15-32% in the Mediterranean basin and by 29-53% in South America (Klausmeyer and Shaw, 2009).

Many studies have been published addressing GHG emissions in Mediterranean cropping systems. These works contributed to characterize soil processes responsible for GHG emission in these systems, indicating possible pathways for GHG mitigation. There is a need, however, to synthetize this knowledge in order to identify distinct GHG emission patterns from soils of Mediterranean cropping systems. At the same time there is also a need to integrate this knowledge into more comprehensive Life Cycle Assessment GHG emission models considering all the relevant processes involved in the C footprint of Mediterranean products, and to discuss their broader implications in terms of climate change adaptation and other socio-environmental impacts.

Chapter 2. Objectives

General objective

To characterize greenhouse gas (GHG) emissions in Mediterranean cropping systems in order to identify emission hotspots and the mitigation potential of alternative management practices, with a special emphasis on organic farming and associated practices.

Specific objectives

Studies 1 and 2: to characterize major soil processes leading to GHG emissions in Mediterranean cropping systems, as influenced by management practices: soil N_2O emissions (Study 1) and carbon sequestration (Study 2)

Studies 3 and 4: to assess the total global warming potential (GWP) of representative Mediterranean cropping systems, as influenced by management practices: herbaceous crops (Study 3) and woody crops (Study 4)

Subspecific objectives

Study 1.

(1) To compare and contrast direct N_2O emissions and emission factors from the application of organic and synthetic fertilizers, describing the influence of agricultural practices and environmental factors;
(2) to get an overview of indirect N_2O sources related to fertilizer use, both upstream and downstream of the cropping system;
(3) to identify options for mitigating N_2O emissions, and their respective drawbacks, through the water management and application of organic fertilizers in these agroecosystems;
(4) to detect the main sources of uncertainty and research gaps in currently available information.

Study 2.

(1) To estimate the mean change in soil organic carbon (SOC) content and SOC sequestration rate associated to the adoption of recommended management practices (RMPs) and to the application of different C input rates;
(2) to analyze the effect of different organic farming systems and practices on C sequestration;
(3) to detect the main sources of uncertainty and research gaps in currently available information.

Study 3

(1) To determine the influence of organic management on area-based and product-based greenhouse gas (GHG) emission balance of a range of herbaceous crops representing organic production in Spain;
(2) to identify critical processes in the carbon footprint of organic and conventional Mediterranean herbaceous cropping systems and crop products;
(3) to analyze the effect of N_2O emission factor choice and coproduct consideration (allocation and system expansion) in the estimation of the carbon footprint of Mediterranean herbaceous crop products;
(4) to identify critical options for improving the carbon footprint of organic and conventional Mediterranean cropping systems and crop products.

Study 4

(1) To determine the influence of organic management on area-based and product-based GHG emissions in a range of fruit tree orchards representing organic production in Spain;
(2) to dentify critical processes implied in the global warming potential of organic and conventional Mediterranean fruit tree orchard products;
(3) to analyze the effect of carbon sequestration rate calculation and coproduct consideration in the estimation of the carbon footprint of Mediterranean orchard products;
(4) to identify critical options for improving the carbon footprint of organic and conventional Mediterranean fruit tree orchard products.

Chapter 3. Theoretical and methodological framework

The theoretical and methodological bases of this dissertation are discussed in this chapter. Section 3.1 outlines the main theoretical grounds, from ecology to agroecology and climate science. Section 3.2 focuses on the methodology employed in studies 1 and 2, the meta-analysis, while life cycle assessment, the methodology employed in studies 3 and 4, is discussed in Section 3.3.

3.1 The biophysical study of agriculture

The study of agriculture from a biophysical perspective involves many scientific disciplines and shifting research paradigms, which are overviewed in this section. First, I overview the theoretical basis of the biophysical study of socio-ecosystems as complex systems, with a particular focus on social metabolism (Section 3.1.1). Then, I describe the role of agroecology in providing an analytical framework for the application of social metabolism to the specific case of the study of agricultural sustainability (Section 3.1.2). Last, I discuss some historical hits in the history of climate science and its application to agricultural GHG mitigation, as one component of agricultural sustainability (Section 3.1.3).

3.1.1 Addressing the complexity of socio-ecosystems to advance towards sustainability

As described in Chapter 1, climate change mitigation in agriculture is part of the major challenges related to achieving sustainability in a context of global change and resource depletion, with multiple interlinked processes interacting through feedbacks to generate non-linear responses. These challenges cannot be faced from the classical reductionist perspective which has long prevailed in science (Nguyen and Bosch, 2013). The reductionist approach fails to understand non-linear responses of complex systems and the interactions between them. Indeed, the integration of the knowledge generated within and among different scientific disciplines in order to face the sustainability problems has become one of the major challenges of science. As a result, new paradigms emerged based on the ideas of complexity, transdisciplinarity, ecology and sustainability. The ecological paradigm adopts a biocentric perspective and combines epistemology (post-normal science) and ethics (principle of responsibility) into ideas such as the "precautionary principle" (González de Molina and Toledo, 2014). These ideas are incorporated into new transdisciplinary research fields such as environmental history, ecological economics, social ecology, political ecology and agroecology.

An important basis of these new perspectives lay in the ideas of systems and complexity. With antecedents such as general systems theory (von Bertalanffy, 1968), complexity science has been classified into different types of complexity which are often complementary (Manson, 2001). **Algoritmic complexity** would include information theory, being the latter widely applied (Ulanowitz, 2001), but also criticized for equating data with knowledge (Manson, 2001). **Deterministic complexity** deals with chaos theory and catastrophe theory, employs deterministic mathematics and attractors, which are values towards which a system variable tends to settle, allows for dynamic behavior using feedbacks (Section 1.1.1), and includes sensitivity to initial conditions (the "butterfly effect") and bifurcations (sudden shift from one attractor to another) (Prigogine and Stengers, 1984). Last, **aggregate complexity** studies how individual elements relate among themselves to create complex systems, paying attention to the relationships between elements and how it defines the internal structure of the system, the environment which continuously interacts with the system, learning and memory, which implies that its history shapes the system allowing adaptations to the changing environment, emergent qualities (the capacities of a complex system are greater than the sum of its constituent parts), and change through three processes: self-organization, dissipation and self-organized criticality (Manson, 2001). Systems and complexity theories are being increasingly adopted in many fields of natural sciences. In biology, they are applied at all scales (Kitano, 2002, Odum, 1983), from molecules, to cells, organisms and ecosystems. Within ecology, they are applied to the study of food webs (Dunne et al., 2002, Gravel et al., 2011), mutualistic networks (Bascompte et al., 2006, Rohr et al., 2014) and host-parasitoid networks (Ings et al., 2009). They are also essential in modern climate science (Sections 1.1.1 and 3.1.3). In social sciences, complexity theory has also been applied widely, from the conceptualization of social institutions as complex systems (Ostrom, 1990) to applications in economic theory (Schweitzer et al., 2009) or governance (Dietz et al., 2003).

From a systems perspective, ecosystems are considered complex adaptative systems that dissipate energy in order to compensate for the law of entropy (Jørgensen and Fath 2004). Society and nature are two complex systems whose interactions in coupled human and natural systems, or **"socioecosystems", are complex systems themselves**, showing typical behaviors such as nonlinear dynamics, reciprocal feedback loops, time lags, resilience, resistance, heterogeneity, and "surprises" (Liu et al., 2007). These features lead to abrupt responses when the changes trespass a threshold and the system shift to an alternative state (Scheffer et al., 2001). It is therefore necessary to adopt a systemic and transdisciplinary perspective for the study of the interactions between society and nature, and particularly for the study of sustainability. This way, new approaches to sustainability "include changing the focus from seeking optimal states and the determinants of maximum sustainable yield (the MSY paradigm) to resilience analysis, adaptive resource management, and adaptive governance" (Walker et al., 2004).

The specific study of the interaction of nature and society from a biophysical viewpoint is addressed by **social metabolism**. The concept of social metabolism, or the idea of applying the biological concept of metabolism to social systems and their interaction with natural ones, was already used by Marx, who talked about "metabolism between man and nature as mediated by the labour processes". After a long dormancy period, important advances were made in this field in the 1970s and 1980s, but the concept of social metabolism was not really re-launched until the 1990s with the works of Fischer-Kowalski and collaborators (e.g. Fischer-Kowalski and Haberl, 1993), in which social metabolism was presented as a useful framework to study the physical input-transformation-output processes between societies and their natural environments. After a wide expansion and application of the concept in the following two decades (e.g. Haberl et al., 2004, Krausmann et al., 2009, Barles, 2009, Kennedy et al., 2015), González de Molina and Toledo (2014) further developed the concept to incorporate the social dimension (cultural and symbolic), represented by flows of information, thus reconciling the ecological with the social applications of complexity theory.

An important idea which was later incorporated into socio-metabolic research is Georgescu-Roegen (1971, 1976) distinction between two distinct elements of the production process: the "**fund elements**" (e.g. land, livestock, labor, machinery) transform input energy and material "**flow elements**" into qualitatively different energy and material outputs, while maintaining their physical identity and integrity through the production process. This framework allows linking the socio-metabolic processes with sustainability, as some flows are needed to maintain the quality of fund elements, while other damage them.

The main **methodological frameworks** of social metabolism are the "material and energy flow accounting" (MEFA) (Haberl et al., 2004) and the "multi-scale integrated analysis of societal and ecosystem metabolism" (MuSIASEM) (Giampietro et al., 2009), which incorporates the fund-flow ideas. Other methodologies are also related to social metabolism, such as life cycle assessments (LCA) (Section 3.3), input-output analyses (IO) (e.g. Wiedmann et al., 2007), human appropriation of the net primary production (HANPP) (Imhoff et al., 2004, Haberl et al. 2004a, Krausmann et al., 2013), ecological footprint (and carbon, nitrogen, water, etc.) (Wackernagel et al., 2002, Haberl et al. 2004a), nutrient budgets (GRAFS) (Billen et al., 2015), or energy balances (EROIs) (Murphy and Hall, 2010).

Five stages can be identified in the social metabolism: appropriation, circulation, transformation, consumption and excretion (González de Molina and Toledo, 2014). From a social metabolism viewpoint, resource depletion problems would be related to the "**extraction**" or "**appropriation**" stage of the socio-metabolic process, while most global change problems, particularly as emissions of GHG and other pollutants to the land, water and air, would be related to the "**excretion**" stage of the socio-metabolic process. Thus, GHG emissions are excretion flows of the social metabolism, which negatively affect the quality of the fund "climate". The use the planetary boundaries (Section 1.1.1) as references to construct sustainability indicators (Fanning and O'Neil, 2016) is a possible approach to measure the impacts of socio-metabolic processes on the quality of global "natural funds" such as the climate.

3.1.2 Applying the socio-metabolic approach to agriculture

Despite the linkage of social metabolism, and systems dynamics, with agroecology has not been theoretically formalized yet, we believe that they are naturally complementary, and they actually overlap in some aspects. In fact, agroecology could be considered as the branch of social metabolism devoted to the study of agro-ecosystems and food systems.

Agroecology definitions vary widely, from the first ideas focused on "the application of ecology to agriculture" (Wezel and Soldat, 2009), to the concept of "ecology of food systems" (Francis et al., 2003). Despite today we are far from a consensus, we can remark some of the main elements of published definitions (Altieri, 2002, Dalgaard et al., 2003, Wezel et al., 2009, Caporali, 2010, Gliessman, 2012, Méndez et al., 2012), saying that agroecology refers to either a scientific discipline, a social or political movement, or an agricultural practice, which deals with agricultural sustainability challenges from a transdisciplinary, participative and transformative approach, first incorporating the ecological knowledge to agricultural management, and afterwards integrating these productive aspects with environmental, social, economic and ethical issues. Agroecology thus incorporates the new system paradigm into agricultural science (Caporali, 2010)

If we focus on **agroecology as a science**, Dalgaard et al. (2003) demonstrated that it meets the stablished norms of science to be considered as a scientific discipline: communalism, universality, disinterestedness, originality and doubt. Dalgaard et al. (2003) remarked that agroecology as a science has two main issues to resolve, namely scaling and interdisciplinarity. Scaling issues have to do with the problems associated to the projection of knowledge generated at low spatial scales to higher scales in which measures are applied. Although recent works have already advanced in this direction (see Wezel and Soldat, 2009), there is still a need for a "political agroecology" that incorporate agroecological criteria into the policy-making processes that shape agri-food systems (González de Molina, 2013). This scaling problem has important implications for the implementation of mitigation measures, as discussed in Section 5.3.3. On the other hand, the problems of reconciling different world views through interdisciplinarity is common to all sustainability science, as discussed in Section 3.1.1.

The unit of analysis in agroecology is the "**agroecosystem**", which was defined by Guzmán and González de Molina (2015) as "part of the general metabolism of society, specifically dedicated to the appropriation of the products of photosynthesis. They are often confused with the farm, that is, with the crop (…). However, agroecosystems are coherent units through which biogeochemical flows circulate, with human appropriation thus giving rise to different degrees of intervention". The concept emerged in the 1970s, and was later developed and broadened by Conway (1987), Altieri (1989), Gliessman (1998) or Guzmán et al. (2000) to incorporate ideas such as the dependence on external energy and environmental impacts, and linking it to sustainability through attributes such as productivity, stability, resilience, equity and autonomy. Later on, the concept was situated within socio-metabolic research and interpreted from a thermodynamic perspective (Guzmán and González de Molina, 2015, Guzmán and González de Molina (Eds.), In Press). The incorporation of complexity and thermodynamics to the concept of agroecosystem diversification as a major pillar of the agroecological idea of agricultural sustainability. Following this line, the main advantages of diversified systems can only be seen when the full range of functions and ecosystem services is taken into account (Kremen and Miles, 2012).

The interdisciplinary approach of agroecology has been widely applied to the study of **agricultural sustainability** (Tomich et al., 2011), and also to food systems sustainability (Guzmán et al., 2013). Specific methodologies have been developed incorporating multiple criteria and participatory research, such as the Framework for Evaluation of Natural Resource Management Systems (MESMIS) (Masera et al., 1999), which have also been applied to Mediterranean systems, particularly *dehesas* (Nahed et al., 2006, Ripoll-Bosch et al., 2013, Escribano et al., 2014). The results of these studies show the usefulness of this multi-criteria and participatory approach (Astier et al., 2011, 2012), while also indicating that these framework methodologies could be improved with tools such as simulation models, linear programming and trade-off analysis (Speelman et al., 2007).

Thus, an agroecological approach, able to deal with agroecosystem complexity, is needed in studies addressing any aspect of agricultural sustainability, even if they are focused on just one component of sustainability, such as GHG mitigation. On the other hand, synthesis efforts are needed for being able to elaborate tools for a feasible but precise assessment of GHG emissions within broader multi-criteria and multi-actor studies. **Agroecology applications to the challenges of climate change** in agriculture have recently been proposed, particularly in the field of adaptation to climate change (Duru et al., 2012, Roge et al., 201, Altieri and Nichols, 2015). The study of agricultural GHG emissions from an agroecological perspective, however, has received much less attention.

3.1.3 Climatic science and agricultural GHG emissions

The first linkages between **GHG emissions and climate** were made in the 19th century, with studies about the absorption of infrared radiation by water vapor, hydrocarbons, methane (CH_4) and CO_2 (Fourier, 1827). Before the end of the century, Arrhenius (1896) estimated that cutting atmospheric CO_2 by half would lead to an ice age, while doubling it would lead to a warming of

5-6ºC. In the second half of the 20th century, many other works appeared relating atmospheric CO_2 and climate change (e.g. Moller, 1963, Schlesinger, 1984), and linking fossil fuel combustion to rising CO_2 concentration and temperature (Rotty, 1974). In the following decades, the incorporation of complexity features to climate models, and particularly their linkage with ecosystems and socio-ecosystems, put forward the risks associated to non-linear, abrupt and irreversible responses, lowering the safety thresholds (Higgins et al., 2002, Hansen et al., 2013). After the creation of the Intergovernmental Panel on Climate Change, which led to comprehensive periodic reviews of the state of the art of climate science, the evidence supporting the idea of the anthropogenic contribution to climate change has been increasing further (IPCC, 1990, 1996b, 2001, 2007, 2013). In 2010, 255 members of the National Academy of Science of the USA signed a letter published in the journal Science claiming that, despite the inherent uncertainty of all scientific research, the evidence supporting that climate is warming due to human-related greenhouse gas (GHG) emissions, with potentially very severe socio-economic impacts, justified an urgent action from policymakers and the public to address the causes of climate change (Gleick et al., 2010).

An essential component of GHG mitigation in agriculture is carbon sequestration. However, as many other functions of SOM, this function was overlooked for a long time by soil scientists. Manlay (2007) and Feller et al. (2012) identify three stages in the **evolution of SOM concept**: i) The *humic* period: Up to 1840 plants were thought to take organic matter from the soil. This was systematized in the "humus theory" of Thaer (1809), which advocated for the use of organic fertilizers to feed the plants; ii) The *mineralist* period: From 1840s to 1940s mineral fertilization expanded supported by applications of Sprengel's and Liebig's advances in mineral plant nutrition (van der Ploeg et al., 1999) and by the progression of urbanization (which made more difficult to reuse waste) and fossil fuel use; iii) The *ecological* period: From 1940s, the functions of SOM (Section 1.2.3) were discovered and gradually incorporated into agronomic research. The work of Balfour (1944) linked soil health with plant and human health, incorporating a holistic view of agriculture that was the basis of the philosophy of **organic farming**. Last, the relationship between soil carbon and climate was not acknowledged until very recently: the work of Trumbore et al. (1996) linking temperature rise with soil carbon loss, and the first estimations of the magnitude of soil carbon in the global carbon cycle (Schlesinger et al., 2000) were followed by many studies recognizing the contribution of SOM to climate change mitigation and adaptation (see Sections 1.2.3 and 5.2.2).

The acknowledgement of the contribution of agriculture to climate change (Section 1.2) has led to the development of international protocols for the assessment of GHG emissions. The **assessment of agricultural GHG emissions** is commonly performed following IPCC guidelines (IPCC, 1996a, 2006). The **IPCC** offers a hierarchical approach for the estimation of GHG emissions by parties to the United Nations Framework Convention on Climate Change

(UNFCC). Tier 1 is the default methodology, and can be used to estimate GHG emissions with minimum data requirements, employing global or regional emission factors derived from literature. Higher tiers involve the use of local emission factors or more detailed modelling. Most countries estimate soil GHG emissions in their national GHG inventories using IPCC Tier 1 methodology, although some of them are applying more advanced methods (Lokupitiya and Paustian, 2006). IPCC default values (Tier 1) for the calculation of **N_2O emission**s are emission factors referred to the amounts of N inputs applied to soils in the form of synthetic or organic fertilizers. These factors are world average values that might be accurate in certain situations: for example, re-estimating N_2O emissions from Chinese croplands using local emission factors and direct measurements led to emission factors similar to Tier 1 IPCC factors (Gao et al., 2011). In other situations, these emission factors can be far from field measurements (Gerber et al., 2016, Cayuela et al., Under Review, see also Section 1.2.2 and Study 1).

The structuration of emissions information by economic sectors in **National Inventories of GHG** does not acknowledge for the relationships between these sectors, makes very difficult to have a clear picture of total emissions involved in a certain activity and to determine the influences of mitigation measures on emissions in other economic sectors or in other countries. In the particular case of agriculture and food consumption, this mainly implies that, on the one hand, emissions associated to input production (Section 1.2.4), are omitted as they are allocated to different sectors or countries, and, on the other, it is not possible to discriminate emissions from crop production or food consumption, as the agro-food chain involves activities in virtually all sectors. Moreover, the IPCC methodology for soil carbon accounting is very coarse because of the high uncertainty, and most countries do not report emissions associated to this process.

These problems suggest that the assessment on agricultural GHG emissions, and particularly of mitigation options, should be performed using climate-specific emission factors and C sequestration rates. An effective way of obtaining those coefficients is through systematic reviews (Section 2.2). Then, these coefficients can be integrated into full GHG emission balances following integrated approaches such as LCA (Section 2.3).

3.2 Systematic reviews

3.2.1 Concept

Science is constructed using empirical studies as bricks to test the hypotheses about how nature works. However, as we have seen, nature is a network of complex systems in which multiple factors are always implied in a given observation, which hampers achieving generalizable conclusions. In fact, it has been argued that most scientific findings are false (Ioannidis, 2005) as long as the existence of significant results in a restricted universe do not imply the general prevalence of that effect. Thus, obtaining true results would only be possible with very large studies covering a representative enough universe (Ioannidis, 2005, Moonesinghe et al., 2007). This is an impossibility in most of the complex situations of the real world, but it is possible to get closer to it through meta-analyses integrating results of multiple individual studies in order to observe general trends

The **qualitative, descriptive or classical review** is a technique employed to synthetize and resume the scientific knowledge in a given research field, so that the most relevant knowns and unknowns are highlighted. Thus, reviews aim to summarize the latest findings and point to research needs. They also help to inform about the findings to non-experts in the field. Borenstein et al. (2009) underline a number of problems associated to the use of descriptive review as a tool to synthetize knowledge in a research area. For example, they lack a statistical support to claimed findings. They also have to exclude many references for space reasons, which is usually done with an absence of selection criteria.

The absence of selection criteria can be a possible source of **bias** in descriptive reviews. For instance, significance of studies is a common source of bias: this can be done by the journal (journals tend to select studies with significant results) or by the author of the review (which is common in descriptive reviews). On the other hand, studies published in higher-impact journals have more chances to be included in the review. But it may occur that high-impact and low-impact journals tend to publish different types of results, for example lower-ranked journals might be more prone to publish non-significant results.

Simple techniques for quantitative reviews can overcome some shortcomings of qualitative reviews. They include vote counting and combined probability methods. **Vote counting** is the oldest and simplest method for quantitative review, and it is usually employed to support claims in descriptive reviews. The results are summarized into three categories: statistically significant in the expected direction (+), statistically significant in the not expected direction (-) and statistically non-significant. Votes are summed to give an easily understandable result. This basic scheme can be refined with procedures such as the sign test, in which the result is contrasted in a binomial distribution. Confidence intervals can also be calculated taking into account the effect size. Moreover, other techniques can be used to detect bias. Some shortcomings of the vote counting method are that it yields usually too conservative results, preventing the identification of the existence of an effect when sampling size or effect size are small. Another problem is that it does not provide information about the size of the effect, only about it existence. Another important limitation is that it ignores the fact that the absence of significance does not mean an absence of effect, as it can be caused by the small size of the sample. **Combined probability methods** combine research results based on exact probability values (or their transformations). They imply an improvement over vote-counting methods because they consider the study size, but they do not provide information about the effect size. There are different types: minimum p method, logarithm sum, Z sum, p sum.

Meta-analytical methods are focused on the calculation of a general effect based on the combination of the effects of the individual studies. The core concept is the effect size, which refers to the magnitude of the impact of one variable over another (Cohen, 1969). Unlike conventional statistical methods, meta-analyses perform a weighting of individual studies, usually based on their variance. Other alternative methods to combine effect sizes are based on Bayesian methods. The next sections describe meta-analytical methods with detail.

3.2.2 Steps of meta-analyses

Design step includes defining study objectives, scientific hypothesis to be tested, effect size metrics and additional variables to be studied. Along the process we will find many situations where bias may occur. This bias can be identified through quantitative methods, as explained below.

Identification of relevant studies is generally performed employing reference databases such as the Web of Knowledge, Web of Science or SCOPUS. Keywords have to be defined, and complementary methods can be used, such as searching the reference lists of the initial set of papers identified.

Selection of which data should be combined and whether they should be combined is an important step of the meta-analysis process (Lau et al., 1997). In principle, the criteria defined in the "Design" step should be applied in this one, although we may modify those criteria at this steps in order to take into account new information we had found in the reviewed studies. Some of the selection criteria are: i) whether the study include enough information for analysis (i.e. information to calculate effect size, or variability information such as standard deviation or standard error); ii) the study design (i.e. laboratory studies only or controlled greenhouses or only field studies only); iii) the year of study, if technology or typical dosing changes (i.e. when a new regulation may alter the characteristics of a given group); iv) the dosage used in the study (i.e. include only studies with realistic N application rates); v) the language of the article (it may not be possible to understand the article); vi) The minimum sample size - very small studies may be unrepresentative and/or not worth the effort; v) The source of publication. The meta-analysis can be limited to papers indexed in SCI, or include also grey literature or other sources. This should be specified in the criteria. **Bias** can be graphically studied with funnel plots (Egger et al., 1997), in which precision metrics are plotted against effect sizes, although funnel plot asymmetry should not be equated with publication bias (Sterne et al., 2011). There are some techniques to analyze publication bias: Rosenthal equation can be used to answer the question: how many more studies are needed to avoid publication bias? And Orwin equation can be used to calculate the number of missing studies needed to move the effect size to the value it would have without bias.

Abstraction refers to data compilation. It is important to minimize possible errors in the publications, in the interpretation of the results by the meta-analysis researcher, or during data entry or data organization. Some practices to overcome these errors are comparing abstract, text and figures of reviewed publications looking for inconsistencies, use 2 or more independent reviewers, or reporting the result of the data abstraction. In this sense, even when errors are not detected in a published meta-analysis, it is interesting to be transparent publishing the raw database with the collected information. The quality of the studies can be assessed in this phase, stablishing criteria and verifying which studies meet them.

Analysis phase usually include studying heterogeneity and applying a model for the calculation of the mean effect size (or effect sizes). Additional analyses may include sensitivity analyses, in which we test the effect of changing some assumptions or study compositions. Analysis step is explained in detail in Section 3.2.4.

Presentation and interpretation of results is the final step, in which all the relevant information obtained during the meta-analysis process is integrated. The most usual graphical representations are "forest plots", which display effect sizes and their confidence intervals (Anzures-Cabrera and Higgins, 2010). Then we have to interpret the significance of the calculated combined effect in view of the information we have about our database, such as the quality of the data or the heterogeneity. We also have to discuss the biological meaning of our findings, their possible explanations and their possible implications.

3.2.3 Effect size

As stated by Hedges (2008), effect sizes are quantitative indexes of the relationships between variables found in research studies, which can provide an understandable summary of research findings, and also allows combining findings from multiple studies in order to get cumulative evidence. Hedges (2008) warns against the use of p-values as an effect size metric, because p-values do not represent the size of an effect, but only the statistical significance, which depends on the sample size. Thus, p-values provide information about the reliability of a relationship found in an experiment, but not on the magnitude of that relationship.

Hedges (2008) identified 3 broad effect size types: the standardized mean difference family, the standardized regression coefficient family, and the odds ratio family. An additional effect size metric, which is used very often in ecology and agronomy studies, is the response ratio (Hedges et al., 1999). The **standardized mean difference, or Cohen's d** (Cohen, 1977) refers to the difference between the means of the two studied groups, divided by their combined standard deviation. Thus, it naturally represents "how false" the null hypotheses are in quantitative terms, and it has various advantages such as being independent from sample size and from scale, which is very valuable in meta-analyses (Hedges, 2008). **Odds ratios** are appropriate effect sizes for dichotomous data, and a complete set of statistical procedures has been developed to conduct all types of meta-analyses based on odds ratios (Haddock et al., 1998). The **response ratio** is widely used in ecology and refers to the log proportional change in the means of a treatment and control group. Some works have further developed variance metrics of response ratios, helping to quantify very diverse experimental designs with this effect size (Lajeunesse, 2011).

3.2.4 Data analysis methods

The fixed effects model and the random effects model are two very common types of meta-analysis. They have been described in research papers (Hedges and Vevea, 1998, Borenstein et al., 2010) and text books (Borenstein et al., 2009). Other types of meta-analysis are distribution-free meta-analyses, that employ non-parametric approaches based on bootstrapping for the calculation of confidence limits, and Bayesian meta-analysis (Higgins et al., 2009). Furthermore, in the last years some progress has been made on the development of multivariate meta-analysis methods, which extend the classical meta-analysis to multiple outcomes (Jackson et al., 2011a, Mavridis and Salanti, 2013) and network meta-analysis, which analyze more than one treatment effects on one outcome (Salanti et al., 2014).

In the **fixed effect model** we assume that there is a single common effect (a "true effect") in all studies of our population, this is, that the factors that might be influencing this effect are the same. This way, the differences between the effect sizes we find would be due to sampling error only. Thus, with a large enough sample size in each study, the differences between studies would disappear and the effect size of the individual studies would be equal to the true effect size.

Studies with large sampling errors (wide normal distribution curves) have more chances to be far from the true effect size than more precise studies (Borenstein et al., 2009). In practice, we combine individual studies operating the opposite way, by calculating the mean of all studies weighted inversely to their variance:

$$W_i = \frac{1}{V_{Y_i}}$$

Where W_i is the weight of study *i*, and V_{Y_i} is the variance of the effect size Y of that study. That way, the weighted average of all studies would be calculated as:

$$M = \frac{\sum_{i=1}^{k} W_i Y_i}{\sum_{i=1}^{k} W_i}$$

The variance of the mean effect size is estimated as the reciprocal of the sum of the weights:

$$V_M = \frac{1}{\sum_{i=1}^{k} W_i}$$

The standard error, confidence intervals Z and p values can be calculated using these statistics as in a regular statistical analysis.

The assumption of a single effect for all studies made in the fixed effect model might not be sustained in many real-world situations. This is especially true in ecological research (Gurevitch and Hedges, 1999), where complex systems with very variable situations are studied.

In the **random effects models** we acknowledge this fact assuming that there may be different true effect sizes underlying different studies, and that they are normally distributed around a grand mean. In this approach, the observed effect Y of a given study is given by the grand mean, the deviation of the study's true effect from the grand mean, and the deviation of the study's observed effect from the study's true effect (Borenstein et al., 2009). That is,

$$Y_i = \mu + \varsigma_i + \varepsilon_i$$

Thus, to perform a random-effects meta-analysis we compute study variance taking into account not only within-study variance (V_Y), as in the fixed-effects meta-analysis, but also between-study variance (τ^2). The latter parameter can be calculated in many ways, for example the method of moments, or DerSimonian and Laird (1986) method, is one of the most common ones, although it has been criticized, and other methods such as the Q-profile have been proposed (Veroniki et al., 2016).

The variance, standard error and confidence intervals will be larger with a random-effects model than with a fixed-effect model, because the between-study variance is incorporated. It can be argued that the most interesting thing of the random-effects model is not the calculated effect size or significance of the results, but rather to question the sources of the variability, especially when the variance is large.

A first step before the selection of the model is the study of the **heterogeneity** of the population. The statistical test Q helps deciding whether to choose a fixed effects or a random effects model, while other statistics have also been developed to measure heterogeneity in meta-analysis, such as H, I-2 and R (Higgins and Thompson, 2002). The main criteria, however, would be our knowledge about the characteristics of the studied systems. In ecology and agronomy we will most usually find situations when the random effects model is more appropriate. Moreover, the Q statistic has a low statistical power (Borenstein et al., 2009), so even when the significance criteria are not met by the statistic, it can be appropriate to use a random-effects model.

Non-parametric, distribution-free approaches overcome data gaps in the primary studies. In particular, information related to variability (standard deviation, standard error or variance) is often absent in many ecological studies. Sometimes even if these information is not provided in the paper, it is directly provided by the authors after request. In some cases, we can overcome these gaps with estimations. For example, these estimations can be based on p-values, if they are known. The reliability of the estimation would be much lower if we only know the level of significance, or even worse if we do not know the significance at all. On the other hand, many agronomy and ecology studies employ very small sample sizes. We have to take into account that when we statistically analyze a sample with parametric methods we make assumptions about the distribution of our population. In small samples these assumptions might not be verified. In those cases, the use of non-parametric methods could be preferred, because we do not have to assume a normal distribution for the data of our studies. Making fewer assumptions has the advantage of yielding valid results in situations when assumptions of parametric models are not met, and the disadvantage of lowering the precision of the results when they do (Adams et al., 1997).

One of the weighting procedures which maintain the philosophy of meta-analysis of giving more weight to larger studies is the sample size-based weighting (Hedges and Olkin, 1985, Adams et al., 1997)

$$w_{ij} = \frac{N_{ij}^{E} N_{ij}^{C}}{N_{ij}^{E} + N_{ij}^{C}}$$

Another option when variability data is missing is performing an **un-weighted** meta-analysis. This procedure is getting common in the ecology field (e.g. Guo et al., 2002, Tonitto et al., 2006 Berthrong et al., 2009, Nave et al., 2009, Rui et al., 2010).

The basic tools that allow the non-parametric calculation of the distribution of mean effect sizes are the resampling, or **bootstrapping, methods** (Adams et al., 1997). These methods can also be applied to the estimation of missing data (Efron, 1994), and they are based on the creation of multiple subsamples composed of the same number of studies but randomly selected. A new population is created with its own distribution, from which we select the lower and upper limits of our confidence intervals (Campbell and Torgerson, 1999).

The conventional method for the calculation of **bootstrap confidence limits** is the percentile bootstrap (Efron, 1979). In this method, *i* studies with replacement?? are chosen for each class, and a weighted mean effect size is calculated. After repeating this process several times (typically, 1,000-10,000 times), the output values are ordered sequentially, and the lowest and highest 2.5% values are chosen as the bootstrap confidence limits (Adams et al., 1997). A further development of this approach are the **bias-corrected percentile confidence limits**, which could correct for distributions when >50% of the bootstrap replicates are larger or smaller than the observed value, which is common with small samples (Efron, 1987). The resampling tests usually result in wider confidence limits that those obtained with parametric approaches (Adams et al., 1997). Summarizing, the bootstrapping methods have three major practical advantages: they are conservative, they do not make assumptions about parametric distributions, and they allow including a larger number of studies, as they do not require as much information.

3.2.5 Structuring data

Categorization for subgroup analysis is a very common practice in meta-analysis. We make it when we want to compare the performance of different subgroups. Grouping can be based in different characteristics of the subjects (the studied systems, in our case, e.g. a maize vs a wheat field), in different variants of the intervention (different types of treatments, such as no-tillage vs reduced tillage), or in different types of outcomes (e.g. long-term vs short-term effects). In this case possible interaction between independent factors have to be discussed. The basic procedure for a categorized meta-analysis is to perform an independent meta-analysis for each one of the studied groups (Borenstein and Higgins 2013). In the first place, we make a meta-analysis of all data, and independent meta-analyses of all groups. We can then check whether the groups are different between them by determining if their confidence intervals overlap. A more formal way to test these differences is an approach similar to the usual ANOVA. We calculate heterogeneity between groups ($Q_{between}$) and within groups (Q_{within}). The sum of both is the total heterogeneity (Q_{total}). We can test the significance of all these parameters knowing the degrees of freedom. We can also test the significance with Monte-Carlo simulations, which are much more robust and do not imply making assumptions about the sampling distribution.

With **meta-regression** we study the effect of continuous variables on our effect size. In most situations, the most appropriate model for meta-regressions would be the random-effects model, in order to take into account not only within-study variance but also between-study variance. An appropriate assessment of heterogeneity is paramount in meta-regression studies (Higgins and Thompson, 2004)

Complex data structures occur when there are multiple subgroups in the studies. This complexity can be handled in meta-analysis, and different procedures can be applied for each situation: i) **independent subgroups within a stud**y. In those cases, if we do not need to compare subgroups we can either treat the subgroups as separate studies, or use studies as unit or analysis by combining subgroups of the same study, or recreate the summary data for the full study. On the other hand, if we want to compare subgroups we can use subgroups as unit of analysis or we can compare subgroups within the study to estimate a single effect size for that study. A comprehensive explanation of these methods is provided in Borenstein et al. (2009); ii**)** **multiple outcomes or time-points within a study**, i.e. different studies may have more than one response variable, for example the response might be studied at different time points for the same treatment in the same location. In those situations, we cannot treat the subgroups as independent studies, because we would overestimate the precision of the summary effect (if the correlation between the subgroups is positive) or underestimate it (if it is negative). We can overcome this situation by combining effects across different outcomes, taking into account the correlation between the subgroups; iii) **multiple comparisons within a study** comprise situations when a single control group is used for multiple treatment groups within a study, and therefore these effect sizes are not fully independent, as they share a control group. One option is to compute a combined effect size of treatments sharing a control group, while, if we want to compare the treatments, we can remove control groups and compare them directly.

3.3 Life Cycle Assessment (LCA)

3.3.1 Concept

Life cycle assessment (LCA) and similar methodologies aim to address the complexity of modern value chains by quantifying all impacts that can be attributed to, or are consequence of the production of a particular product or activity. The life cycle approach takes into account all emissions involved, not only direct emissions. The exhaustiveness of LCA allows the detection of emission hotspots and a meaningful comparison between mitigation practices, because the possibilities of emission leakages are greatly diminished. In fact, the ranking of mitigation options can be very different when calculated with LCAs and when calculated with partial methodologies such as the IPCC method, because the second ones do not include some important processes of the production chain (O'Brien et al., 2011). Therefore, LCAs are more appropriate for assessing GHG mitigation options from an integrated point of view.

The concept of life cycle assessment (LCA) appeared in the 1960s, while many methodological aspects were developed in the 1970s (Finnveden et al., 2009). In that period, some studies extended the philosophy of energy balances, which were currently popular, to a wider scope of environmental impacts. In the following years, many companies applied this approach to assess the impact of their activities. However, the heterogeneity of methodologies hindered the credibility of the results, preventing their acceptance within the scientific community. This way, LCA did not become clearly stablished in scientific spheres until the 1990s, when an important standardization effort was done (Guinee et al., 2011). In the 21st century, the role of LCA as a policy assessment tool expanded, together with the consequential approach for allocating environmental loads instead of the attributional approach (McManus and Taylor, 2015).

Today we could **define LCA** as an integrative, internationally standardized methodology to quantify environmental impacts of defined systems (products, organizations, lifestyles, territories) along their whole life cycle, this is, "from cradle to grave" (Hellweg and Mila i Canals, 2014). This accounting methodology considers the impacts associated to production, use and residue disposal. The environmental impacts are classified in impact categories, such as climate change, acidification, eutrophication, or ecotoxicity. In each of these categories, the impacts are expressed as one common unit, such as kg CO_2-equivalents, kg SO_2-equivalents, or kg PO_4^{3}-equivalents, respectively.

The methodological framework of LCA is defined by the **International Organization for Standarization (ISO)**, which stablishes the minimum requirements the studies have to meet in order to provide comparable and contrastable results. ISO is composed by 163 members, each of which represents one country (e.g. in Spain it is AENOR) and have right to one vote. This organization was founded in 1947 to set industry standards and facilitate international trade. In that sense, its role in the food production field is controversial, because it can turn into a mechanism to consolidate and legitimate international trade and the bureaucratic control of agrarian production, such as in the case of the European Union regulation on organic farming (Cuéllar, 2011). In the particular scientific context, however, the existence of a common and transparent methodological framework allows realizing directly comparable studies with which to identify the most effective ways to reduce the environmental impact of products, including food. This standardization, however, does not prevent from the existence of specific problems which solution requires some subjectivity, such as the selection of system boundaries (Suh et al., 2004). Another particularly controversial issue is the allocation of environmental burdens between different coproducts of a single production process (Weidema, 2000, Weidema and Schmitz, 2010). As we will see, crop products show additional difficulties for standardizing impact assessment, due to the uncertainty in the estimation of some processes such as soil GHG emissions. A number of approaches can be adopted for these estimations within the LCA framework. Thus, the methodology is flexible enough to be able to adjust it to every particular condition, but, in this field, the resulting methodological heterogeneity may question the comparability of results.

3.3.2 Stages of a LCA

The main stages of an LCA are the goal definition and scoping, inventory analysis, impact assessment and interpretation of results. These stages are interlinked, so that the results obtained in one of them can modify the methodological approach to be taken in the others. This is represented in Figure 3.1.

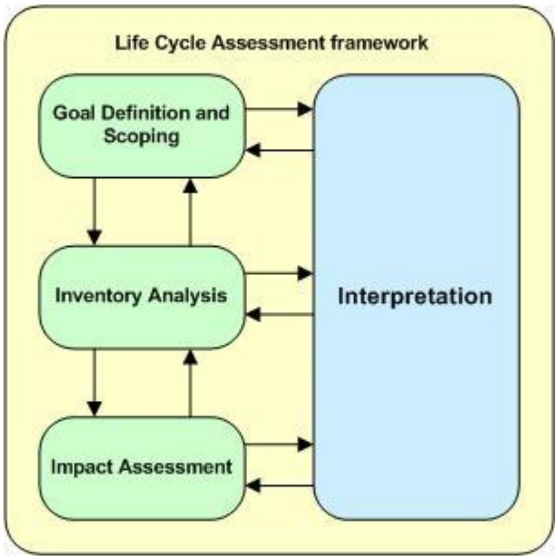

Figure 3.1 Stages of a life cycle assessment (LCA). Source: ISO (2006)

The **goal definition and scoping** is usually considered the most important stage of a LCA (Roy et al., 2009), as it defines the procedures and boundaries of the study. First, the motivations and relevance of the study are set, and the goal is clearly defined. The **system** associated to the product, process or service is described, specifying the **boundaries** of the system, this is, specifying which particular processes will be included. This description is usually accompanied by a diagram showing the main elements and fluxes included in the system. A **functional unit (FU)** has to be chosen to be used as a reference unit to express the results in a normalized way. Its choice depends on the impact categories studied and the study objectives. This unit usually refers to the mass of the product (e.g. kg or Mg), although other units such as energy or surface units may also be applied. The goal definition implies the choice between the two main types of LCA: attributional, which aims to attribute environmental impacts to the functional unit, or consequential, which analyzes the consequences of a change in production. Last, the assumptions which will be made are specified. These assumptions include, in the case of the attributional LCA, the choice between different allocation methods.

The **inventory analysis** stage involves quantifying all flows (inputs and outputs) of materials, energy and emissions of all relevant processes included in the studied system (usually hundreds of them). This is usually the most time-consuming stage, as it implies gathering the primary information about the studied system, which usually requires making interviews or other similar procedures. It is important to collect primary information about the processes that are involved in the life cycle and which particular characteristics have in our case (for example, which fertilizers and how many kg of each type are applied). Then, the inventory information about those processes can be gathered from life cycle assessment databases such as Ecoinvent (in the former example, the materials and energy required to produce one kg of the different fertilizers; http://www.ecoinvent.org/).

The **life cycle impact assessment** (LCIA) involves estimating the environmental impacts based on the inventory analysis and the objectives. A large variety of impacts can be analyzed. There is a core set of environmental impact categories, including energy consumption, climate change, eutrophication, acidification and toxicity (human and ecological toxicity). Climate change category is almost identical to the so-called carbon footprint (Hellweg and Mila i Canals, 2014), which is a member of a wider family of "footprints", such as the ecological, water or energy footprints (Fang et al., 2014, Hoekstra and Wiedmann, 2014). The core set of LCA impacts has been complemented by other environmental impact categories, such as water and land use, and by social indicators, such as social wellbeing. The preliminary step in the LCIA is the **classification**, which involves grouping the results of the inventory analysis among the impact categories selected in our assessment. Then, it takes place the **characterization**, in which the contribution of each process to each impact category is assessed. In this step, the potential impacts are expressed in a comparable way, for example converting N_2O and CH_4 fluxes to CO_2-equivalents. The following steps are optional according to ISO standards, which means that they are not available in all methods implemented in LCA software packages. Through **normalization**, the category indicator is compared by a reference (or normal) value. This can be used for making the relevance of the results easier to understand, and it also allows comparing impact categories between them. Another way to compare impact categories is **weighting** across impact categories, which involves multiplying impact category results by weighting factors to create a single score.

The last stage is the **interpretation of results**, in which the results are evaluated and interpreted to identify the possibilities for reducing the environmental impacts of the studied systems. Potentially, recommendations could be derived about the system design aimed to reduce environmental impacts.

3.3.3 The multi-product problem defines LCA types

Many production systems generate more than one product. In those cases, we have an **allocation problem**, because it is necessary to stablish criteria to allocate environmental loads. We can distribute loads through **partitioning**, by allocating a percentage of the load to each product based on economic, biophysical or other criteria. Another approach is **system expansion**, in which the specific functions of the coproducts are taken into account. With system expansion we assume that, if we have a main product and a coproduct, the coproduct would be displacing another product in the market, so the environmental loads of the displaced product are allocated to the coproduct, while they are subtracted from the total system loads to obtain the loads of the main product. System expansion is commonly not considered an allocation method, but rather a method to avoid allocation (Weidema, 2000, BSI, 2011).

The allocation method is closely linked to the study objectives. In **attributional** LCAs (ALCAs) we aim to determine the environmental loads that can be attributed to certain products, and thus we commonly use partitioning, while in **consequential** LCA (CLCAs), in which the effects of the changes are assessed, system expansion would be applied (Table 3.1). ALCAs have been usually regarded as **retrospective** LCAs, as they study systems as they are, while CLCAs have been usually regarded as **prospective** LCAs, as they are focused on the consequences of changes.

Table 3.1. Some characteristics of attributional and consequential LCAs. Sources: Tillman, 2000, Guinée et al., 2002, Thomasen et al., 2008, Earles and Halog, 2011

	Retrospective/Attributional	**Prospective/Consequential**
Approach	Status quo	Change-oriented
Goals	Accounting. Reflect causes of the system	Study consequences of changes (scenarios consstruction)
System limits	Additivity. Completeness	Affected parts of the system
Allocation procedure	Partition. Expansion is optional	Expansion. Partition not used
Data choice	Average	Marginal ? (at least partly)
System subdivision	-	Foreground and background
Required knowledge	Physical mechanisms	Physical and market mechanisms (but less processes studied)
Boundaries	Static processes	Processes affected by changes in demand
Quality	Less sensitive to uncertainty	More sensitive to uncertainty

In spite of this general pattern, which is verified in most situations, Weidema (2003) pointed out that attributional LCAs can be prospective, and consequential LCAs can be retrospective (Table 3.2).

Table 3.2 Relationship between the categorizations retrospective/prospective and attributional/consequential. Source: adapted from Weidema (2003)

	Attributional	Consequential
Retrospective	Allocate responsibility to past actions (Who should we blame about how things are?)	Casual explanation of past actions (What would have happened if we had or had not done that?)
Prospective	Allocate responsibility to future actions (Who should we blame about how things will be?)	Casual explanation of consequences of future actions (What would happen if we do or do not do that?)

In a **consequential LCA** we would ask what would be affected by a change (usually small) in demand (Dalgaard et al., 2005). Thus, the consequential approach involves the integration of physical models employed in attributional LCAs with economic models assessing market behavior (Earles and Halog, 2011). The usual approach to integrate these two modeling methods is based on partial equilibrium modeling, while a heuristic approach to determine affected technologies has also been used (Earles and Halog, 2011). In recent years we could see the application of more complex models, such as the Multi-Market, Multi-Regional PE Models and Computable General Equilibrium models, as well as the incorporation of other economic concepts such as rebound effects and experience curves (Earles and Halog, 2011).

PAS 2050:2011 regulation (BSI, 2011) stablishes a **hierarchy in allocation methods** depending on the type of system, the available information and the study goals. According to this recommendation, the preferred method for allocating loads between coproducts is distinguishing the specific processes that are involved in each of them within the production process, and thus make independent models for each coproduct. When this is not possible, as it is usually the case, system expansion should be tried, if it is possible to identify a product which is displaced by the coproduct and stablish environmental loads representing that product. If this were not possible, partitioning following economic criteria could be applied. In some specific situations, PAS 20150:2011 recommends using physical criteria such as mass or energy, for example when the co-products have the same functions despite having different prices, such as different types of apples. The results of a study applying both the attributional and the consequential approaches (Ekvall and Andrae, 2006) suggest that these two LCA methods provide complementary information about the environmental impacts of production processes. An interesting discussion about the **implications of ALCAs and CLCAs in the assessment of GHG mitigation** practices took place in the Journal of Industrial Ecology in 2014. Plevin et al. (2014a) criticized the use of ALCA to assess the GHG mitigation potential of a given practice or technology, arguing that it does not consider the consequences that the expansion of such a technology would have in the real world. They consider that consequential LCAs could overcome this problem, despite they could also increase uncertainty with the construction of scenarios. Brandao et al. (2014) responded supporting CLCAs, arguing that omitting the study of effects do not reduce the uncertainty of ALCAs, despite reducing the statistical error. In their view, ALCAs would be more precise (smaller statistical error), but they would also be less accurate (more prone to bias), while CLCAs would be closer to reality, this is, more accurate, despite being less precise. Brandao et al. (2014) propose applying a postnormal approach for the construction of consequential scenarios, implying social actors in a participative process. Dale and Kim (2014) also responded to Plevin et al. (2014a), arguing that consequential LCAs, as they are being applied, do not improve the estimations because they are based on unproved scenarios. For example, consequential LCAs only model the consequences of biofuel expansion, not those of the continued use of the fossil fuels which they substitute. Last, Hertwich (2014) questioned the validity of substitution assumptions by CLCAs models, pointing out that the expansion of alternative technologies does not necessarily reduce the use of standard technology. In their response to the comments, Plevin et al. (2014b) claim that complexity and uncertainty have to be acknowledged and incorporated in the analysis in order to produce results more useful for decision-making.

3.3.4 Applying LCA to GHG accounting in agriculture

LCA is the most complete and standardized methodology for assessing the carbon footprint of agricultural products. The holistic approach allows the identification of emission hotspots along the production chain, this is, those processes where mitigation efforts would be more effective. This approach is also very appropriate for comparing different production methods, as it makes possible to identify unintended effects of practices focused on a specific process. This way, possible sources of leakage can be identified and avoided (Finnveden et al., 2009, Knudsen et al., 2011). Moreover, this methodology also allows to compare the carbon footprint with other environmental impacts of agricultural production, assessing possible tradeoffs of mitigation practices beyond GHG emissions. These advantages, together with the development of analytical tools within LCA and the increasing interest in the assessment of the climatic impacts of food and biofuel production, have boosted the number of LCA studies of crop products (Bessou et al., 2012, Ruviaro et al., 2012)

The **boundaries of LCAs of food products** vary widely depending on study objectives, and they rarely follow a "cradle to grave" approach (which involves waste disposal and human excretion in our case), but this might not be necessary for comparing individual food products. The "cradle to grave" approach has been applied to the study of full agri-food systems or dietary patterns (e.g. Muñoz et al., 2010). Many LCAs of food products stablish system boundaries from "cradle to shop" (e.g. Beccali et al., 2010, Ardente et al., 2006), while others restrict the boundaries from "cradle to agro-industry gate" (e.g. Aranda et al., 2005, Avraamides et al., 2008, Gazulla et al., 2010, Salomone et al., 2012), including from the production of agricultural inputs up to packaging the food product. When the aim is to focus on the agricultural production process, the boundaries are stablished from "cradle to farm gate" (e.g. Sanjuán et al., 2005, Flysjo et al., 2011).

The application of LCA methodologies is usually focused on marketable products or processes (e.g. Nemecek et al., 2012 for crop products, or De Vries et al., 2010 for livestock products), although there is an increasing number of examples of wider research subjects, ranging from average individual diets (e.g. Muñoz et al., 2010) to whole economic sectors such as the livestock sector (Lesschen et al., 2011) or the whole country level (Hellweg and Mila i Canals, 2014).

Despite its advantages, the application of LCA methodology to agricultural and food products can be very controversial. On one hand, all common problems of LCAs can also be encountered when studying agricultural products. In fact, these **problems are worsened by the complexity of agricultural systems**. In these systems, the choice of the LCA type (attributional or consequential) and the allocation method or the product substitution model become critical. Allocation problems can be found in production systems generating secondary products, such as meat production in milk (Cederberg and Stadig, 2003, Thomassen et al., 2008, Kristensen et al., 2011) or egg (Pelletier et al., 2014) production systems, straw in cereal production (Powlson et al., 2008), manure in livestock production systems (Mogensen et al., 2014), or agri-food industry multi-product systems such as oil and cakes (Thomassen et al., 2008, Reinhard and Zah, 2009). An additional problem is that many of the co-products of multi-product systems are also inputs of other agricultural production processes (for example, manure as fertilizer, or straw and cakes as feed), making loops that can easily lead to circular reference problems and generating allocation problems not only in the output side, but also in the input side of the production process.

Other systems, such as crop rotations or polycultures, produce **multiple products**, often sharing inputs between them, making difficult to separate each crop as an independent production system. In those cases, PAS2050-1:2012 recommends allocating environmental burdens following simplified criteria: i) emissions produced during the first year after application are allocated to the crop planted that year; ii) emissions produced during the following years are evenly shared between all other crops in the rotation. However, crop rotations are composed by species that may benefit from each other, such as cereal benefiting from nitrogen in legume crop residues (Nemecek et al., 2008), which would make the analysis even more complex.

Another important source of uncertainty is the estimation of **biogenic GHG emissions** from soils and animals, particularly emissions of trace gases (N_2O and CH_4) and the ecosystem carbon balance. The technological components of agricultural LCAs are usually well characterized (e.g. Kim and Overcash, 2003, ecoinvent Centre, 2007) and show relatively low variability, Therefore, they can be modelled using general databases such as Ecoinvent, although regional differences might exist, and they should be taken into account (Aguilera et al., 2015). On the contrary, biochemical processes, such as those taking place in soils and in animal bodies, are very variable and dependent on the specific conditions. In spite of this, the usual approach in LCAs for the estimation of these components of the carbon footprint is to employ Tier 1 emission factors from IPCC guidelines for national emission inventories (IPCC, 1996a, 2006). As we saw in Section 3.1.3, tier 1 N_2O emission factors are general, world-average factors that may not be very accurate in specific situations such as Mediterranean climate cropping systems.

The uncertainty in the estimation of **carbon sequestration** is even larger than that in the estimation of N_2O emissions, which has favored the omission of soil carbon dynamics from agricultural GHG balances, despite this process has been identified as the one with the largest GHG mitigation potential within agriculture (Section 1.2.3). Very few studies have aimed to integrate carbon sequestration within full agricultural GHG balances. Some examples of these exceptions are the works by Kim and Dale (2005), Adler et al. (2007), Halberg et al. (2010), Brandao et al. (2011), Knudsen et al. (2014), Venkat (2012) or Godard et al. (2013). These studies show that the contribution of soil carbon dynamics to the total GHG balance cannot be overlooked. Some authors have proposed analytical methods for the inclusion of soil carbon balance in LCA models, both as a component of the GHG emission budget (Petersen et al., 2013) and as a standalone indicator of soil fertility status, which is used as a proxy for life support functions (Milá i Canals, 2007) or biotic production potential (Brandao et al., 2013). The estimation of carbon sequestration within LCA studies can be done with several methods, which can be hierarchically classified in three types (Milá i Canals, 2007, see also Section 2.1):

(a) **Direct Measurements**: this approach yields the most accurate results, but it is usually prohibitively expensive for application to LCA. LCA practitioners, however, can benefit from already existing SOC databases with site-specific information.

(b) **Model calculations**: many process-based models of soil biogeochemistry can estimate SOC stock changes, together with the emission of trace gases such as N_2O and CH_4. These models require more or less detailed information of soil properties and climatic parameters during the study period, but it is usually much more easily available than direct measurements of SOC evolution. Therefore, this type of method has been applied more often in LCAs (e.g. Kim and Dale, 2005)

(c) **Estimates from literature values**: many databases provide spatially-specific information on soil properties, including global (e.g. Hengl et al., 2014, Shangguan et al., 2014, Stockmann et a 2015, Jones and Thornton, 2015) regional (e.g. Jones et al., 2005 for Europe), national (e.g. Chiti et al., 2012 for Italy, Rodríguez-Martín et al., 2016 for Spain) and subnational (e.g. Schulp et al., 2009, in the Netherlands, Rodríguez-Lado et al., 2015 for Galicia, NW Spain) levels. These assessments, however, usually reflect the current situation but they do not provide enough information for the estimation of carbon sequestration rates as a response to changes in management practices or land uses. For that purpose, the IPCC (2006) and other sources (e.g. Milá i Canals, 2007) provide typical carbon sequestration rate values for common management practices and land use types. These values, however, are usually very gross approaches, representing world or global region averages, and not considering important parameters such as the amount of carbon inputs applied to the soil.

Therefore, there is a need to develop **simple methodological approaches** to estimate soil GHG fluxes as a response to management changes, in order to integrate soil carbon dynamics in LCA. These approaches should require only basic information on agro-climatic characteristics of the study area, as this information is usually limited in LCA studies. Thus, estimates from literature values are probably the most feasible approach for integrating soil processes in LCA, but these estimates would be much more accurate if they were based on climate-specific data.

Chapter 4. Publications

The four publications included in this chapter (Studies 1-4) are independent research works. Therefore, they have their own sections and they do not refer to other sections in the dissertation. They also have their own reference lists.

4.1. Study 1

The potential of organic fertilizers and water management to reduce N_2O emissions in Mediterranean climate cropping systems. A review.

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Eduardo Aguilera.[a,b,*]; Luis Lassaletta[c,e]; Alberto Sanz-Cobena[d]; Josette Garnier[e]; Antonio Vallejo[d]

[a]*Spanish Society of Organic Farming (SEAE), Camí del Port, S/N. Edif ECA Pat Int 1º - km 1 (Ap 397) 46470 Catarroja (Valencia, Spain)*

[b] *Universidad Pablo de Olavide, Ctra. de Utrera, km. 1, 41013, Sevilla (Spain)*

[c] *Department of Ecology, Universidad Complutense de Madrid, c/José Antonio Novais s/n 28040 Madrid, Spain*

[d] *Escuela Técnica Superior de Ingenieros Agrónomos, Universidad Politécnica de Madrid, Ciudad Universitaria, 28040 Madrid, Spain*

[e]*UPMC/CNRS, UMR Sisyphe, Box 123, 4, Place Jussieu, 75005 Paris, France*

* *Corresponding author: estudioscc@agroecologia.net, +34 913365652*

Abstract

Environmental problems related to the use of synthetic fertilizers and to organic waste management have led to increased interest in the use of organic materials as an alternative source of nutrients for crops, but this is also associated with N_2O emissions. There has been an increasing amount of research into the effects of using different types of fertilization on N_2O emissions under Mediterranean climatic conditions, but the findings have sometimes been rather contradictory. Available information also suggests that water management could exert a high influence on N_2O emissions. In this context, we have reviewed the current scientific knowledge,

including an analysis of the effect of fertilizer type and water management on direct N_2O emissions.

A meta-analysis of compliant reviewed experiments revealed significantly lower N_2O emissions for organic as opposed to synthetic fertilizers (23 % reduction). When organic materials were segregated in solid and liquid, only solid organic fertilizer emissions were significantly lower than those of synthetic fertilizers (28 % reduction in cumulative emissions). The EF is similar to the IPCC factor in conventionally irrigated systems (0.98 % N_2O-N N applied^{-1}), but one order of magnitude lower in rainfed systems (0.08 %). Drip irrigation produces intermediate emission levels (0.66 %). Differences are driven by Mediterranean agro-climatic characteristics, which include low soil organic matter (SOM) content and a distinctive rainfall and temperature pattern. Interactions between environmental and management factors and the microbial processes involved in N_2O emissions are discussed in detail.

Indirect emissions have not been fully accounted for, but when organic fertilizers are applied at similar N rates to synthetic fertilizers, they generally make smaller contributions to the leached NO_3^- pool. The most promising practices for reducing N_2O through organic fertilization include: i) minimizing water applications; ii) minimizing bare soil; iii) improving waste management; and iv) tightening N cycling through N immobilization. The mitigation potential may be limited by: i) residual effect; ii) the long-term effects of fertilizers on SOM; iii) lower yield-scaled performance; and iv) total N availability from organic sources. Knowledge gaps identified in the review included: i) insufficient sampling periods; ii) high background emissions; iii) the need to provide N_2O EF and yield-scaled EF; iv) the need for more research on specific cropping systems; and v) the need for full GHG balances.

In conclusion, the available information suggests a potential of organic fertilizers and water saving practices to mitigate N_2O emissions under Mediterranean climatic conditions, although further research is needed before it can be regarded as fully proven, understood and developed.

Keywords: Nitrous oxide; Mediterranean cropping systems; synthetic fertilizer; organic fertilizer; irrigated crops; rainfed crops

Contents

1. Introduction

9. References

1. Introduction

Nitrous oxide (N_2O) is a powerful greenhouse gas (GHG). It is 298 times stronger than CO_2 at the 100-year time horizon and in 2005, it accounted for 6.1% of combined GHG radiative forcing (Forster et al., 2007). According to the cited authors, the atmospheric N_2O concentration rose from 270 to 319 ppb in the period 1900-2005, after having previously remained relatively stable for the previous two millennia. Agricultural emissions represent about 60% of global anthropogenic N_2O emissions. They increased by 17% from 1990 to 2005 and are projected to increase by 35-60% up to 2030 (Smith et al., 2007).

Nitrogen fertilizer applications to soils, whether organic or synthetic, result in N_2O emissions, as this gas is a by-product of the transformation of N compounds added to the soil. N_2O fluxes from soil are mainly driven by microbial activity, through nitrification and denitrification processes (Firestone and Davidson, 1989). In spite of the existence of a large body of knowledge on the mechanisms that underlie these pathways, there is still an insufficient understanding of the finer details of the process, such as how the composition of organic N fertilizers affects denitrification, nitrification and emission rates (Vallejo et al., 2006).

Besides soil emissions after fertilizer applications (direct emissions), fertilizer-related N_2O production can also result from indirect emissions. Downstream of the cropping system, N_2O is produced when N compounds, and particularly leached nitrate (NO_3^-) and volatilized ammonia(NH_3), are subsequently transformed into N_2O (IPCC, 2006a). These indirect sources can represent a significant fraction of total agricultural N_2O emissions (Garnier et al., 2009). Upstream of the cropping system, N_2O and other GHG are emitted as by-products of fertilizer production, storage and transport (Snyder et al., 2009). Although these emissions are very dependent on the methods used to obtain fertilizers, half of synthetic N fertilizer-related GHG emissions could occur in the production phase, whereas the other half occurs from the soil (Tirado et al., 2010). In 2001, fertilizer production accounted for 1% of the global energy demand; 72% of this energy corresponded to N,and a further 16% to compound fertilizers containing N (Ramírez and Worrell, 2006).

There is increasing interest in the application of organic fertilizers to soils (e.g, Hargreaves et al., 2008, Petersen et al., 2003, Singh et al., 2008, Smil, 1999), as they can contribute to climate change mitigation through C sequestration (Diacono and Montemurro, 2010), at the same time helping to tackle problems associated with waste management and meeting the nutrient and organic matter needs of agricultural soils (Tirado et al., 2010). Nevertheless, the use of organic fertilizers also has a number ofdrawbacks, which include the energy costs associated with transport and the land spreading of the fertilizers (Wiens et al., 2008), potential pollution with heavy metals and other toxic substances (Petersen et al., 2003), the availability of organic N sources, and GHG emissions (Snyder et al., 2009).

The type and composition of fertilizers have been shown to affect direct N_2O emissions from cropped soils (Stehfest and Bouwman, 2006), although the differences between applying organic and synthetic fertilizers are still not clear. For example, Laegreid and Aastveit (2002) analyzed several databases and found higher direct N_2O emissions from manure than from mineral fertilizers. Although there is great uncertainty in the estimation of the effect of fertilization type on indirect N_2O emissions, overall emissions could actually be slightly lower for organic fertilizers, as calculated in a top-down analysis (Davidson, 2009).

These information gaps are especially relevant for Mediterranean-type cropping systems, where an increasing body of knowledge is being built. However, the data involved are rather dispersed in the existing literature and no compilations have yet been made. Most N_2O emission studies have been conducted under temperate climatic conditions, but ecological processes in regions with Mediterranean climates are affected in different ways to those subject to other climatic conditions, as has been shown in various fields of research including plant physiology (González-Fernández et al., 2010), N biogeochemistry (Breiner et al., 2007), limnology (Álvarez-Cobelas et al., 2005) and biodiversity (Barriga et al., 2010). Aschmann (1973) defined areas with Mediterranean climates as those in which at least 65% of annual rainfall occurs in winter and in which annual precipitation ranges from 275 to 900 mm. The average winter temperature is below 15ºC, but the number of hours per year with temperatures below 0ºC does not exceed 3% of the total. This climate is characterized by seasonal dryness and many of its subtypes could be classified as semi-arid. This climate type is found in five different parts of the world; they are generally on the west coasts of continents and between latitudes 32º and 40º north and south of the equator. These areas are: the Mediterranean basin; California; Central Chile; the Cape region of South Africa; and South and South-West Australia (Fig. 1). The diversity of soil types is very wide in areas with Mediterranean climates on account of their extensions, variety of geological origins and different land uses. The only common feature shared by these soils is their low organic matter content, while they also often contain low levels of mineral nutrients. The Mediterranean biome is highly biodiverse, but subject to extremely intense development pressure (Underwood et al., 2009). Large increases in yield have been achieved through the intensification of agricultural practices in areas with Mediterranean climates (e.g. Ryan et al., 2009). However, the associated agronomic practices are now undermining both soil and water quality (Zalidis et al., 2002) and are also affecting the natural biodiversity through, for example, N deposition in ecosystems (Ochoa-Hueso and Manrique, 2011).

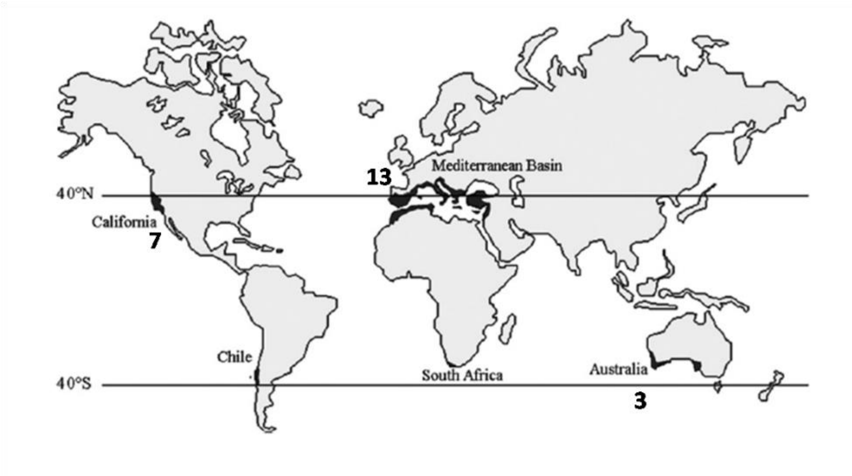

Figure 1. Regions of the world with a Mediterranean climate and number of papers measuring field N_2O emissions in each region.

Research into the effects of N fertilization on N_2O fluxes under Mediterranean conditions has yielded somewhat contradictory results. Moreover, IPCC Tier 1 N_2O emission factors (EF) are mainly based on temperate climate data, and Leip et al. (2011) have recently emphasized the need of more specific EF considering particular parameters such as fertilizer type or climate. The development of Mediterranean climate-specific information on N_2O emission would therefore greatly improve the accuracy of national GHG inventories for these regions. In this context, we have reviewed and analyzed the currently available scientific information associated with fertilizer-related N_2O emissions under Mediterranean climatic conditions, with the following objectives:

1) to compare and contrast direct N_2O emissions from the application of organic and synthetic fertilizers, describing the influence of agricultural practices and environmental factors

2) to get an overview of indirect N_2O sources related to fertilizer use, both upstream and downstream of the cropping system

3) to identify options for mitigating N_2O emissions, and their respective drawbacks, through the application of organic fertilizers in these agroecosystems and to detect the main gaps in currently available information

2. Methods

A wide review was performed which included every paper containing data on N_2O emissions in Mediterranean agroecosystems. Articles were consulted on the ISI Web of Knowledge database by simultaneously typing the words "nitrous oxide" or “N_2O”, "emission", and either the word "Mediterranean" or the name of a country with territory in areas with a Mediterranean climate, according to the map in Fig. 1 This search based on keywords was complemented with a search through the literature cited in the articles found.

N_2O emission data have been expressed as cumulative emissions and EF. Cumulative emission data refer to the sum of N_2O fluxes over the reported measurement period. For measurements covering more than one year, the values cited were converted to one-year references. In some cases, cumulative emissions were estimated through the interpolation of emission values obtained from the figures provided in these papers. Figures were digitized using GetData v.2.24 software (Fedorov, 2002). N_2O EF refers to the proportion of fertilizer N that is released as N_2O-N during the measurement period after discounting emissions from an unfertilized control treatment (equation 1). Studies covering less than a full growth season were excluded from EF analyses. In many cases, measurement period was extended for a full growth season or more, but was shorter than one year; therefore the annual EF may be underestimated. In order to explore different methodologies, we estimated an additional EF (EF_{annual}) and cumulative emissions by linearly extrapolating average N_2O emission levels during sampling period to the rest of the year..

$$\mathrm{EF} = \frac{\text{kg N}_2\text{O-N (F)} - \text{kg N}_2\text{O-N (C)}}{\text{kg N fertilizer applied}} \qquad (1)$$

where EF is the emission factor (N_2O-N emitted as % of fertilizer-N applied) and N_2O-N (F) and N_2O-N (C) are the reported cumulative N_2O emissions (kg N ha^{-1} yr^{-1}) from the fertilized and control (unfertilized) treatments, respectively. N fertilizer applied is the rate of N applied during the study (kg ha^{-1} yr^{-1}). For slow N release organic amendments (i.e. solid organic fertilizers), "available N" was preferentially chosen as the "N fertilizer applied" value instead of the total N content in the fertilizer; this was in line with the procedure followed by most of the authors whose work was reviewed. The "available N" approach takes into account the fact that only a fraction of the N contained in organic fertilizers mineralizes during the measurement period. The EF for organic fertilizers obtained applying this approach is therefore greater than that obtained with the total N approach recommended by the IPCC (2006a). However, this avoids any possible underestimation of the EF associated with the residual effect of solid organic fertilizers, which is particularly useful for short sampling periods (although this assumption is open to criticism, see section 6.1). EF based on total N applied was also calculated for some analyses in order to compare the results obtained with the available N method and then discuss them.

In a first approach, the effect of fertilizer type on cumulative emissions of N_2O and EF was studied using a general matrix containing the results of all the publications reviewed. Fertilizer type was grouped at four levels: i) synthetic, including urea, ammonium nitrate, ammonium sulfate and NPK compound fertilizers; ii) organic (solid), including residues of cover crops (legumes and non-legumes), organic manure, composted municipal solid waste, composted cattle and sheep manure, and composted thick fractions of digested pig slurries; iii) organic (liquid), including raw or digested pig slurries; iv) organic + synthetic, including mixtures of organic and synthetic sources of N, except when the organic N was only represented by residues of the previous crop, in which case the treatment was classified as "synthetic".

The influence of water management on N_2O emissions was studied by classifying this factor in three categories: i) rainfed systems, in which no irrigation water was applied; ii) high water systems, including furrow, sprinkler and micro-sprinkler irrigation; iii) low water systems, including surface and subsurface drip irrigation techniques.

We have observed a high degree of variability in the published N_2O emissions associated with organic and synthetic fertilization. Therefore, in a second approach, the dataset was further narrowed down by restricting the data used to pairwise comparisons of field emissions from organic and synthetic fertilizers in order to examine them meta-analytically. Weighted meta-analysis requires information on variance and the number of replications of each treatment; studies that did not report these data were therefore also excluded. These pairwise comparisons represent experiments where synthetic and organic fertilizers are compared under similar agro-climatic conditions and application rates. In many occasions, multiple organic fertilizers were compared to a common control (synthetic) group. Therefore, in order to avoid an over-estimation of the precision of the mean effect size (Borenstein et al., 2009, Hungate et al., 2009), the resulting pairs were combined into one composite effect size, which variance is given by equation 2.

$$\mathrm{var}\left(\frac{1}{m}\sum_{i=1}^{m}Y_i\right)=\left(\frac{1}{m}\right)^2\left(\sum_{i=1}^{m}V_i+\sum_{i\neq j}(r_{ij}\times\sqrt{V_i}\times\sqrt{V_j})\right) \quad (2)$$

Where m represents the number of correlated pairs, Yi and V_i are the effect size and the variance of pair i, and r is the correlation coefficient, which is equal to 0.5 because all pairs share a common control.

No EF standard deviations (SD_{EF}) were provided in any of the reviewed papers, so they were calculated from cumulative emission standard deviations and sample sizes, following equation 3.

$$SD_{EF}=\frac{\sqrt{\frac{(n_F-1)\times SD_F^2+(n_C-1)\times SD_C^2}{n_F+n_C-2}}}{\text{kg N fertilizer applied}} \quad (3)$$

where n_F and n_c are the number of observations in the fertilized and control (unfertilized) treatments, respectively. SD_F and SD_C are the standard deviations in the fertilized and control treatments, respectively.

We chose the response ratio (RR) as the effect size unit for both cumulative emissions and EF data. This RR-value is the ratio between some measured quantity in the experimental (organic in our case, $\bar{X}_{Org}$) and control (synthetic, $\bar{X}_{Syn}$) groups ($RR\equiv\bar{X}_{Org}/\bar{X}_{Syn}$). We used the natural log of RR (L_i) to perform the analysis (equation 4 for its mean, and equation 5 for its variance), because this transformation results in a much more normal sampling distribution in small samples (Hedges et al., 1999).

$$L_i=\ln(\bar{X}_{Org})-\ln(\bar{X}_{Syn}) \quad (4)$$

$$\mathrm{var}_i=\frac{(SD_{Org})^2}{n_{Org}\times\bar{X}_{Org}}+\frac{(SD_{Syn})^2}{n_{Syn}\times\bar{X}_{Syn}} \quad (5)$$

where SD is the standard deviation and n the number of observations in the organic and synthetic groups. Individual effect sizes were pooled together into a common effect size following a random effects model; this is appropriate for situations in which individual studies are not projected to share a common effect size. This weighted mean effect size and its related statistical parameters were calculated using Comprehensive Meta-Analysis software (Borenstein and Rothstein, 1999). The weighted mean L_i and their variances were transformed back in the presentation of the results.

In order to summarize all the data reviewed, information on cumulative emissions and EF grouped by other factors was also provided: N fertilizer rate, crop type and tillage. Some treatments were excluded from all the analyses: i) Rice paddies, because only one measurement was available (Skiba et al., 2009) and the conditions for N_2O production are very different from those of the rest of the crop types; ii) Experiments employing chemicals or additives such as nitrification inhibitors, because their use is limited on a global scale (Stehfest and Bouwman, 2006); iii) Treatments with N fertilizer rates of more than 300 kg N ha^{-1} $yr^{-1}$of available N, or 500 kg N ha^{-1} yr^{-1} of total N (solid organic fertilizers), as higher rates are not considered representative of typical Mediterranean agroecosystems. We only found one such treatment (in Heller et al., 2010); iv) Studies published before 2000, in order to assure a relatively homogeneous measurement methodology. This implied the exclusion of only one paper (Ryden and Lund, 1980).

3. Direct N_2O emissions in Mediterranean cropping systems

3.1 General overview of the papers reviewed

Published field research into N_2O emissions from Mediterranean agricultural systems began 3 decades ago in the Santa Maria Valley, Santa Barbara, California (Ryden and Lund, 1980), although this pioneering study was not followed by any others until very recently, when research results from the La Poveda research station in Central Spain were published (Vallejo et al., 2005). Over the short period from 2005 to mid-2011, a valuable body of knowledge has been acquired. 24 field experiments are now available in the scientific literature, which relate to 13 different study sites (Table 1). These relate to 3 of the 5 areas in the world that have Mediterranean climates. All the field measurements of N_2O emissions were performed using closed chambers, with a sampling frequency of usually between one day and two weeks, which usually increased after important management events such as fertilization, irrigation or significant rainfall. The measurement frequency was highest in the Australian studies (Barton et al., 2008, 2010, 2011), which used soil chambers connected to a fully automated system and collected samples every 180 minutes. These field data were complemented by 11 laboratory experiments that addressed N_2O emissions from Mediterranean soils, or under specific Mediterranean conditions, such as the addition of organic matter sources from Mediterranean cropping systems (olive residues) (García-Ruiz and Baggs, 2007).

Table 1. Characteristics of field experiments measuring N_2O emissions in Mediterranean cropping systems

Reference	Study site	Crop type	Fertilizer[2]	Irrigation[3]	Duration (days)	Soil type	Observations
Barton et al., 2011	Cunderdin, SW Australia	Lupin	Legume CC, none	Rainfed	350	Natric Haploxeralf and Typic Natrixeralf	
Barton et al., 2010	Cunderdin, SW Australia	Canola	U, none	Rainfed	360	Natric Haploxeralf and Typic Natrixeralf	
Barton et al., 2008	Cunderdin, SW Australia	Wheat	U, none	Rainfed	365	Natric Haploxeralf and Typic Natrixeralf	
Burger et al., 2005	Russell Ranch (LTRAS-CIFS[1]), Davis, California, USA	Tomato	Compost, CC, synthetic fertilizers	Furrow		Typic Xerothent and Mollic Haploxeralf	Not included (no data on cumulative emissions)
De Gryze et al., 2010	Central Valley, California	Various rotations	Compost, legume CC, synthetic fertilizers	Furrow, rainfed			Not included (only simulated N_2O emission reported). Tillage comparisons.
Garland et al., 2011	Arbuckle, California, USA	Vineyard, legume mix (cover crop)	Legume CC, U-AN	Drip (vineyard), rainfed (CC)	194	Willows silty clay	Tillage comparisons
Heller et al., 2010	Volcani, Israel	Maize	NPK, chicken manure	Drip	365	Typic Rhodoxeralf	One treatment with very high N fertilization (not included). Tillage comparisons
Kallenbach et al., 2010	Russell Ranch, Davis, California	Tomato	LCC, NPK, AS	Furrow, drip	365	Reiff loam and Yolo silt loam	
Kong et al., 2007	Russell Ranch (LTRAS-CIFS),	Maize	Compost, legume CC, U-	Furrow	142	Typic Xerothent and Mollic Haploxeralf	Not included (same experiment as Kong et

	Davis, California, USA		AS				al., 2009)
Kong et al., 2009	Russell Ranch (LTRAS-CIFS), Davis, California, USA	Maize	Compost, legume CC, U-AS	Furrow	142	Typic Xerothent and Mollic Haploxeralf	Tillage comparisons
Lee et al., 2009	Yolo County, California, USA	Maize-Sunflower-Chickpea	U-AN, NPK	Furrow	912,5	Myers clay (Entic Chromoxererts)	Tillage comparisons
López-Fernández et al., 2007	La Poveda, Madrid, Spain	Maize	MSW compost, sheep manure compost, PS, U, none	Furrow	200	Typic Xerofluvent	
Lugato et al., 2010	Beano, Udine, Italy	Maize	Not specified	Irrigation	1095	Chromi-Endoskeletic Cambisol	Not included (only simulated N2O emission reported)
Meijide et al., 2007	La Poveda, Madrid, Spain	Maize	PS, DPS, DPS compost, MSW compost, U, none	Furrow	145	Typic Xerofluvent	Nitrification inhibitor used in some treatments (not included)
Meijide et al., 2009	El Encín, Madrid, Spain	Barley	PS, DPS, SS compost, MSW, U, none	Rainfed	335	Calcic Haploxerepts (USDA)- Calcaric Cambisol (FAO)	
Menéndez et al., 2008	El Malagón, Córdoba, Spain	Wheat-Sunflower-Faba bean	AN, none	Rainfed	22, 29	Vertisol (Typic Haploxerert)	Not included (short measurement period). Tillage comparisons.
Petersen et al., 2006	Reggio Emilia, Italy	Wheat-Alfalfa-Maize-Grass	Manure, legume CC, synthetic fertilizers	Irrigation	365		

Ryden et al., 1980	Santa Maria Valley, Santa Barbara, California, USA	Horticultural crops	Manure, legume CC, NPK	Furrow	365		Not included (old study)
Sánchez-Martín et al., 2008b	El Encín, Madrid, Spain	Melon	AN, none	Drip, furrow	140	Calcic Haploxerepts (USDA)- Calcaric Cambisol (FAO)	
Sánchez-Martín et al., 2010a	El Encín, Madrid, Spain	Melon	DPS, none	Drip, furrow	150	Calcic Haploxerepts (USDA)- Calcaric Cambisol (FAO)	
Sánchez-Martín et al., 2010b	El Encín, Madrid, Spain	Onion	Manure, DPS, none	Micro-sprinkler	365	Calcic Haploxerepts (USDA)- Calcaric Cambisol (FAO)	
Skiba et al., 2009	Castellaro, Italy	Rice, Fennel and Maize	Not specified	Irrigation	365	Not specified	Not included (N_2O emission reported only for rice paddies)
Steenwerth et al., 2008	Monterey, California, USA	Vineyard cover crop	CC, none	Rainfed	365	Cumulic Haploxeroll, or Haplic Chenozem	Tillage comparisons
Steenwerth et al., 2010	Monterey, California, USA	Vineyard	U-AN, CC	Drip	33,3	Cumulic Haploxeroll, or Haplic Chenozem	Not included (short measurement period). Tillage comparisons.
Townsend-Small et al., 2011	Irvine and Popoma, California, USA	Maize, horticulture	AP (maize), AN (horticulture)	Rainfed	365	Sorrento loam and clay loam	
Vallejo et al., 2005	La Poveda, Madrid, Spain	Maize	PS, none	Furrow	215	Typic Xerofluvent	Nitrification inhibitor used in some treatments (not included)
Vallejo et al., 2006	El Encín, Madrid, Spain	Potato	PS, DPS, DPS compost, MSW,	Furrow	150	Calcic Haploxerepts (USDA)- Calcaric	Nitrification inhibitor used in some treatments

	U, none		Cambisol (FAO)	(not included)

[1] LTRAS-CIFS: Center for Integrated Farming Systems, formerly known as the Long-Term Research on Agricultural Systems experiment.

[2] Synthetic fertilizers. U: urea; AN: ammonium nitrate; NPK:: compound fertilizers; AS: ammonium sulphate; AP: ammonium phosphate

Organic fertilizers (solid). CC: cover crops; MSW: municipal solid wastes; SS: sewage sludge; DPS compost: composted thick fraction of digested pig slurry

Organic fertilizers (liquid). PS: raw pig slurry; DPS: digested pig slurry (thin fraction)

[3] Rainfed: no irrigation water is applied; High water: furrow and micro-sprinkler irrigation; Low-water: drip (surface or subsurface) irrigation

3.2 Factors influencing N_2O emissions

3.2.1 General effects of fertilizer type and water management

Emissions were highest for slurries ("liquid organic fertilizers", 4.4 kg N_2O-N ha^{-1} yr^{-1} on average, Table 2, Fig. 2a), followed by organic-synthetic mixtures and synthetic fertilizers (respectively, 3.5 and 3.0 kg N_2O-N ha^{-1} yr^{-1}), whereas solid organic fertilizers and unfertilized treatments ("none" group) showed the lowest values (1.7 and 1.8 kg N_2O-N ha^{-1} yr^{-1}, respectively).

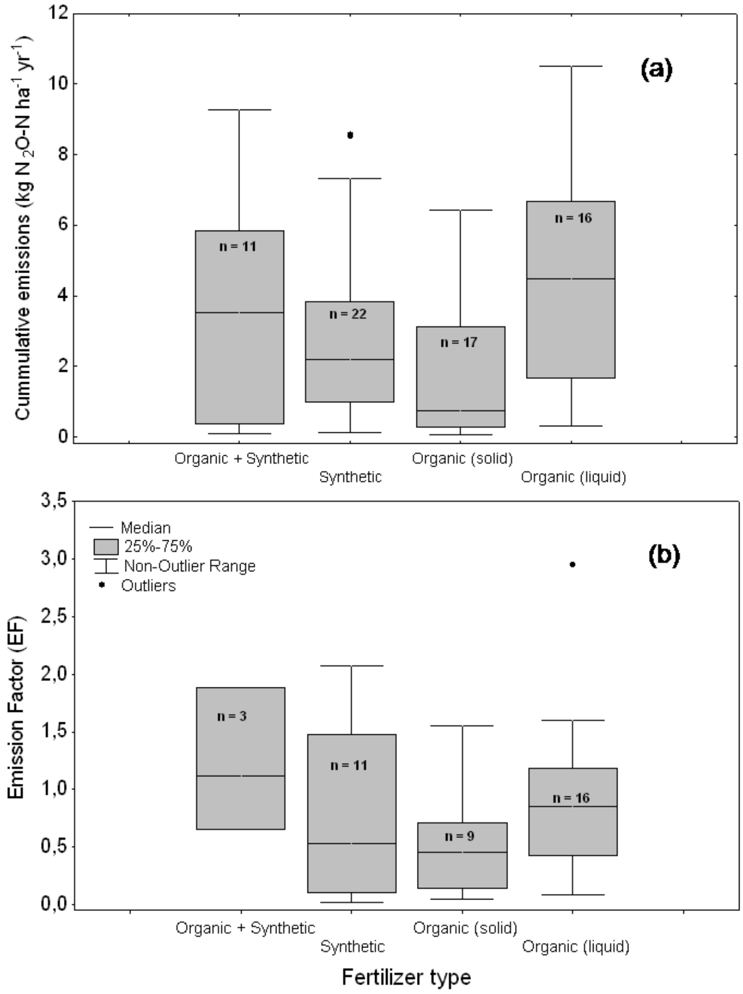

Figure 2. N_2O emissions from Mediterranean cropping systems according to fertilizer type, expressed as: (a) cumulative emissions during experimental period, (b) emission factor. Numbers in the boxes indicate sample sizes.

Despite the high degree of variability in the treatments and conditions included in this analysis, a clear influence of water management on cumulative emissions and EF could be observed in both cases (Fig. 3). Emission levels in rainfed treatments were one order of magnitude lower than those in conventionally irrigated ones (mean cumulative emissions were 0.4 and 4.0 kg N_2O-N ha^{-1} yr^{-1} and average EF values were 0.08 % and 1.01 % for rainfed and high-water irrigation treatments, respectively; Tables 2 and 3). Drip irrigation (low-water category) showed intermediate distributions in emission levels (the averaged cumulative emissions and EF in this category were 1.2 kg N_2O-N ha^{-1} yr^{-1} and 0.66 %).

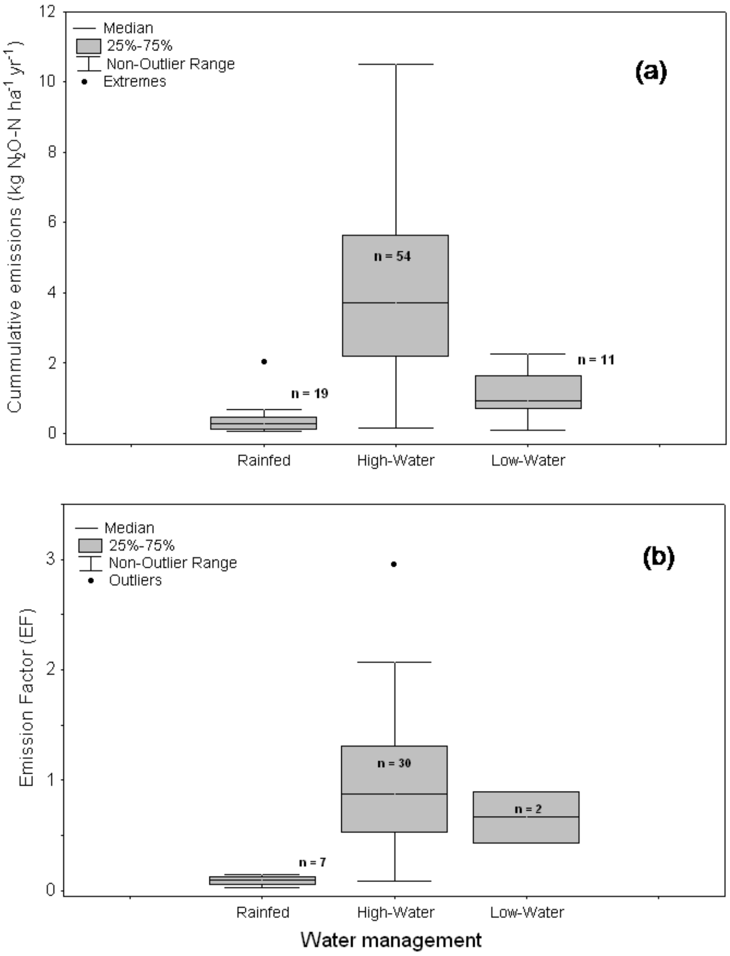

Figure 3. N_2O emissions from Mediterranean cropping systems according to water management type. Rainfed: no irrigation; High-water: conventional irrigation systems (furrow and sprinkler); Low-water: drip irrigation. Emissions are expressed as: (a) cumulative emissions, (b) emission factor. Numbers in the boxes indicate sample sizes.

The high influence of irrigation can be explained by the fact that in Mediterranean agroecosystems this type of management activity is usually applied during the summer dry period, which leads to optimal moisture and temperature conditions for N_2O production (section 3.3). In this way, average cumulative N_2O emissions for Mediterranean cropping systems irrigated with conventional techniques were normally slightly higher than the cumulative emissions for high N application rates (200-250 kg ha^{-1}) obtained from the global data compiled by Stehfest and Bouwman (2006), whereas average EFs were generally similar to the default IPCC factor of 1 %. Other works have also shown that when semi-arid soils are irrigated, an increase in N_2O emissions can be expected (Horvath et al., 2010, Maraseni et al., 2010).

Table 2. Number of observations (N), mean and standard deviation (SD) of cumulative N_2O emissions (kg N_2O-N ha^{-1} yr^{-1}) for some of the factors with a significant influence on N_2O emissions from agricultural fields.

		Cumulative emissions (experiment period)		Cumulative emissions (year estimate)			
	N	Mean	SD	Mean	SD	Mean N fertilizer rate (kg N ha-1 yr-1)	Mean Experiment length (days)
Fertilizer type							
None	18	1.8	1.8	3.9	4.2	0	242
Organic + Synthetic	11	3.5	3.3	6.6	8.0	171	234
Synthetic	22	3.0	2.6	5.2	5.9	159	288
Organic (solid)	17	1.7	1.9	3.1	4.3	147	263
Organic (liquid)	16	4.4	3.0	9.0	6.7	163	220
Water management type							
Rainfed	19	0.4	0.5	0.6	1.0	57	316
High-water	54	4.0	2.6	7.8	6.3	137	231
Low-water	11	1.2	1.0	2.0	1.7	128	254
N fertilizer rate (kg N ha^{-1} yr^{-1})							
0-75	25	1.4	1.7	2.9	3.9	11	248
75-150	16	1.2	1.1	1.2	1.1	113	356
150-225	29	5.3	2.2	11.2	6.0	178	192
225-500	6	2.9	3.0	3.8	3.0	265	291
Crop type							
Horticulture	21	1.9	1.5	3.4	3.4	100	281
Maize	26	4.5	3.2	9.0	7.4	176	207
Other	12	4.0	2.3	8.0	6.3	113	257
Winter cereals	8	0.3	0.1	0.3	0.1	91	343
Vineyard	5	0.4	0.3	0.4	0.2	35	297
Legume	5	0.7	0.9	0.7	1.8	0	220

None	7	3.2	1.5	5.8	2.7	121	221
Tillage type							
Minimum tillage	10	1.9	2.6	2.7	2.8	133	246
Standard tillage	9	1.1	1.4	1.6	1.4	111	232

N fertilizer rate refers to N applied during the experimental period. In the case of solid organic fertilizers this value corresponds to available N in the experimental period, not to total N applied.

In rainfed systems, different limitations on N_2O production are imposed throughout the cropping cycle, especially when the semi-arid conditions are marked. This result was very much in line with low N_2O emissions linked to fertilizer application reported for semi-arid agroecosystems under different climatic conditions (Dick et al., 2008, Galbally et al., 2008). What limited information is available about drip-irrigated systems suggests intermediate emission levels, which points to a high potential for N_2O mitigation when using this water-saving technique (section 5.1). The N_2O mitigation effect of applying low-water irrigation techniques also applies to other world climate types, e.g., under temperate continental (Liu et al., 2011) and arid continental climates (Scheer et al., 2008).

Table 3. Number of observations (N), mean and standard deviation (SD) of emission factor (EF, % N_2O-N over N applied) for some of the factors that have a significant influence on N_2O emissions from agricultural fields.

		EF (experient period)		EF (year estimative)			
	N	Mean	SD	Mean	SD	Mean N fertilizer rate (kg N ha^{-1} yr^{-1})	Mean experiment length (days)
Fertilizer type							
Organic + Synthetic	3	1.22	0.62	3.04	1.57	175	147
Synthetic	11	0.82	0.73	1.71	1.79	142	251
Organic (solid)	9	0.54	0.48	0.97	1.17	147	261
Organic (liquid)	16	0.91	0.70	1.75	1.34	163	220
Water management type							
Rainfed	7	0.08	0.04	0.09	0.05	114	343
High-water	30	1.01	0.63	2.02	1.48	162	213
Low-water	2	0.66	0.33	1.65	0.75	175	145
N fertilizer rate (kg							

N ha^{-1} yr^{-1})							
0-75	1	0.06		0.06		75	360
75-150	12	0.36	0.31	0.37	0.30	115	353
150-225	26	1.06	0.66	2.31	1.42	175	173
Crop type							
Horticulture	10	0.67	0.31	1.10	0.96	136	277
Maize	12	1.33	0.70	2.74	1.34	177	180
Other	6	1.08	0.71	2.61	1.75	158	185
Winter cereals	6	0.09	0.05	0.10	0.05	121	340
None	5	1.08	0.35	0.91	0.64	170	200

N fertilizer rate refers to N applied during the experimental period. In the case of solid organic fertilizers this value corresponds to available N in the experimental period, not to total N applied.

3.2.2. Meta-analysis of the fertilizer type effect

N_2O emission levels from organic fertilizers were in many cases lower than those from synthetic fertilizers when compared on a pair-wise basis, even if the variability was sometimes very high. The mean response ratio for cumulative emissions was 0.77 ($p < 0.05$), meaning that average emission levels were 23% lower for organic than for synthetic fertilizers (Fig. 4a). When organic fertilizers were segregated in solid and liquid ones, only organic solid fertilizer emissions were significantly lower than synthetic (RR = 0.72, $p<0.05$). An EF comparison showed a 23 % reduction for organic fertilizers (RR = 0.77, $p = 0.08$), but the differences were only significant for solid materials (RR = 0.8, $p < 0.05$) (Fig. 4b). When solid organic fertilizer EF was calculated on a total applied N basis, the specific response ratio for this category dropped from 0.8 to 0.37; in other words, from a 20 % reduction in N_2O emissions in comparison with synthetic fertilizers, the reduction reached up to 63 % with total applied N ($p < 0.001$) (Fig. 4c). These results show that the choice between total applied N and available N is highly relevant for EF calculation (section 6.1).

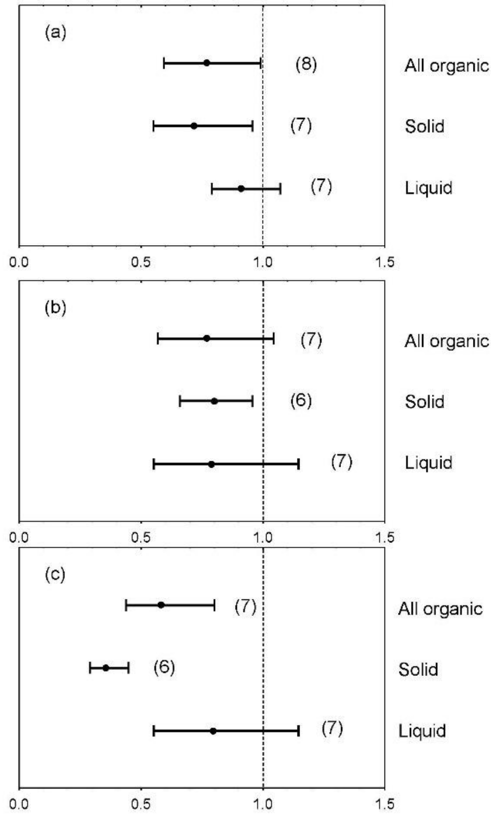

Figure 4. Effect of fertilizer type (organic vs synthetic) on (a) cumulative N_2O emissions; (b) N_2O emission factor (EF) calculated considering available N in solid organic fertilizers; (c) EF calculated considering total N in solid organic fertilizers. The effect is expressed as the response ratio ($RR = \bar{X}_{Org} / \bar{X}_{Syn}$) and categorized into solid and liquid organic fertilizers. Mean values and 95% confidence intervals of the back-transformed response ratios are shown (number of comparisons in parentheses). Emissions are significantly different if confidence intervals do not overlap 1.Number of aggregated paired comparisons are indicated in parentheses (see Methods for independence criteria).

The smaller N_2O emissions from solid compared to liquid organic materials is probably related to their lower concentrations of ammonium (NH_4^+) and their low rate of N mineralization, which usually prevent very high soil mineral N contents being reached. This reduction also seems to be influenced by the semi-arid features which are very common in Mediterranean soils, in which C and N contents are usually low (section 3.3.1). Relatively high emission levels for liquid organic fertilizers (pig slurries) may be influenced by the highly mineralized nature of the N contained in these materials. NH_4^+ levels could actually represent as much as 81

% and 78 %, respectively, of the total N in raw and digested pig slurries (Meijide et al., 2009). Low N_2O emissions for solid organic as opposed to synthetic or liquid organic fertilizers have also been reported in temperate areas (Gregorich et al., 2005).

3.2.3. Other factors influencing N_2O emissions

N application rate

To our knowledge, there were no studies comparing EF as affected by N application rates under Mediterranean conditions. The IPCC approach proposes a linear relationship between N_2O fluxes and fertilizer application rates. However, many experimental data suggest that this relationship may be non-linear, with EF being lowest at low N fertilizer rates and highest at high rates. Accordingly, the global data compiled by Stehfest and Bouwman (2006) suggest a N-shaped curve for N_2O responses to N fertilizer applications, while other studies point to an exponential curve (i.e. Cardenas et al., 2010, Hoben et al., 2011).

In our review, a clear relationship between fertilizer dose and N_2O emissions could be observed in EF (Table 3), but not in cumulative emissions (Table 2), where high-emitting, irrigated control treatments in the lowest dose group (0-75 kg N), and low sample size in the highest dose one (225-500 kg N) may have hidden the underlying trend. 1.8 kg N_2O-N ha^{-1} yr^{-1} was emitted on average from non-fertilized agricultural soils (Table 2). These "background emissions", which occurred on unfertilized control plots, averaged 63.5 % of the fertilized treatment emissions (N=42). Similar results were obtained by Kroeze and Seitzinger (1998) in their top-down analysis; they estimated that in the Mediterranean basins of Europe, 210.7 Mt of background N_2O was emitted from agricultural soils, while only 132.9 Mt of N_2 was associated with the use of manure and synthetic fertilizers. Background emissions are discussed in section 6.2.

Crop type

Although none of the studies analyzed compared emissions from different crop types in the same year, cumulative N_2O emissions varied in a range of from 1 to 4 in function of the crop type assessed in the database analyzed (Tables 2 and 3). The main reason for this was the fact that the influence of crop type is closely related to other important factors that control N_2O production such as the N fertilizer rate and irrigation regime. For example, winter cereals and legumes exhibit exceptionally low N_2O emission levels. Both crops are grown under similar conditions: as rainfed crops during the cool rainy season, with low N fertilizing rates for winter cereals and no fertilizer applications for legumes. Low N_2O emissions have also been observed for cereals in temperate systems (Dobbie et al., 1999). Low N_2O fluxes in vineyards could similarly be related to very low N and water inputs. These systems are rainfed or drip irrigated. In the latter case, the water applied is limited to the soil next to the vine, representing roughly one third of the total surface area (Garland et al., 2011). The highest emissions were registered for maize. N fertilizer application rates and water inputs are normally very high for maize fields, which are cultivated under conventional irrigation techniques and high summer temperatures.

Tillage

The lower mean cumulative emissions registered for tilled as opposed to no-tilled or minimum-tilled treatments, shown in Table 2, were corroborated by the results of most of the studies that specifically addressed this question (Garland et al., 2011, Kong et al., 2009, Lee et al., 2006, Menéndez et al., 2008, Steenwerth and Belina, 2008, 2010), although in some cases the differences were not significant (Garland et al., 2011). There was only one exception to this general trend (Heller et al., 2010), in which emissions were twice as high for the tilled compared to the no-tilled treatment. Tillage events themselves showed no clear effect on N_2O emissions. Thus, whereas in a vineyard soil N_2O pulses occurred after tillage events (Steenwerth and Belina, 2008), in a laboratory experiment there was no N_2O flux response nor any response in the denitrification rate to a simulated tillage event (Calderón et al., 2000). Higher emission rates under minimum or no-tilled treatments were mainly attributed to less aeration, which could have enhanced denitrification, particularly as most of the emissions occurred after irrigation events (Lee et al., 2009, Kong et al., 2009). Studies specifically measuring denitrification show heterogeneous responses of this microbial activity to different tillage practices, ranging from lower (Menéndez et al., 2008) to higher (Melero et al., 2011) denitrification activity under no-tillage, despite both studies were performed under similar conditions (rainfed systems on Vertisol soils). Researchers have also reported higher N_2O emissions from no-tilled plots in both humid and dry climates of the world (Six et al., 2004). In the long term, however, this relationship could disappear or even reverse (Omonode et al., 2011, Six et al., 2004).

Soil Mineral N

Soil mineral N can generally be found either in the form of NH_4^+ or NO_3^-. These two compounds are respectively the substrates of nitrification and denitrification and therefore they both can stimulate N_2O emissions. In accordance with this, some authors have reported a positive correlation between N_2O flux and NO_3^- concentration in the soil (Lee et al., 2006, López-Fernández et al., 2007, Sánchez-Martín et al., 2010b). In some cases, this positive correlation only existed during certain periods (e.g. after the onset of irrigation, López-Fernández et al., 2007) or for certain values of NO_3^- content (e.g. > 6.5 mg N kg$^{\cdot}$ soil^{-1} for temperate climates, Vilain et al., 2010). This last result may explain why many authors have not found any correlation between N_2O emissions and soil NO_3^- (Garland et al., 2011, Ryden and Lund, 1980, Sánchez-Martín et al., 2008b, Steenwerth and Belina, 2008, Vallejo et al., 2005, 2006).

Soil NH_4^+concentration has been significantly and positively correlated with N_2O emissions in many cases (Garland et al., 2011, Heller et al., 2010, Meijide et al., 2007, 2009, Petersen et al., 2006, Sánchez-Martín et al., 2008b, Vallejo et al., 2005). A significant relationship between the NH_4^+ content of the applied fertilizer and N_2O emissions has also been reported (Meijide et al., 2009). In many other cases, however, the correlation is not significant (Barton et al., 2008, 2010, 2011, Steenwerth and Belina, 2008, Vallejo et al., 2001, 2006). There have even been cases in which negative correlations have been found; this occurred in two different experiments involving no-tilled treatments (Garland et al., 2011, Lee et al., 2006). These somewhat contradictory findings suggest that other factors could be limiting N_2O production in cases in which no evident relationship between emissions and soil mineral N has been found. A lack of synchrony between N_2O flux measurement and soil sampling could also have been responsible for some of these results (Barton et al., 2008).

Soil organic carbon (SOC) and dissolved organic carbon (DOC)

In the studies analyzed, SOC concentration was within a narrow range around 1 % of soil mass (7-11.3 g C kg soil^{-1}); as a result, we could not study its relationship with N_2O emissions. DOC is a more dynamic parameter than SOC and its relationship with N_2O emissions was studied in some of the papers reviewed, although they did not show a homogeneous response. Thus, whereas some studies reported a positive correspondence between the two parameters (López-Fernández et al., 2007, Sánchez-Martín et al., 2010b, Vallejo et al., 2006), this relationship was only verified during the irrigation period, and sometimes only at the beginning of this period (Vallejo et al., 2006), while in others, it was absent throughout the experiment (Meijide et al., 2007). In a laboratory experiment, Sánchez-Martín et al. (2008a) showed that labile C (glucose) added to a low-carbon, basic Mediterranean soil strongly reduced N_2O emissions when a mineral

N source was applied. Similar results were obtained with temperate soils and added acetate (Laverman et al., 2010). This effect was not, however, so evident in a high-carbon, acid Scottish soil, where emissions were reduced under high water conditions (90% water filled pore space) but not under low water conditions (40% WFPS) (Sanchez-Martin et al. 2008a). On the other hand, DOC also has an influence on the denitrification rate and N_2O/N_2 ratio, and high-carbon content does not necessarily mean that this substrate is biodegradable for denitrification (see section 3.3.1.).

Water Filled Pore Space (WFPS)

The general association between water inputs and N_2O emissions described in section 3.2.1 is not so clear in the relationship between the WFPS and N_2O fluxes presented in the individual studies. In some cases, N_2O emissions were reported to have increased linearly, with WFPS values increasing from 20% to 70% (Lee et al. 2006), whereas in many other cases this relationship was not found (Garland et al., 2011, Sánchez-Martín et al., 2008b, Steenwerth and Belina, 2008, Vallejo et al., 2005, 2006); this apparently contradicts results reported for temperate climates (Vilain et al., 2010). The lack of correlation between WFPS and N_2O emissions at a temporal scale of days or weeks suggests that WFPS only imposes the upper and lower limits to the microbial processes of nitrification and denitrification; in consequence, within a certain WFPS range, N_2O emissions could be largely unlinked to soil water content (Vilain et al., 2010). Accordingly, increases in WFPS were significantly related to N_2O emission pulses from Mediterranean soils when changes in soil water content were very marked, such as when rainfall or irrigation occurred after a dry period (Barton et al., 2008, 2010, 2011, Garland et al., 2011, López-Fernández et al., 2007, Meijide et al., 2009, Steenwerth and Belina, 2008, see section 3.3 in this paper).

Temperature

The annual temperature range across different sites (13.2-20.9 ºC) was too narrow to divide into categories, so the information was qualitatively reviewed. Temperature is a key factor for microbial activity and is therefore considered to have a pronounced effect on N_2O emissions. Moreover, soil temperature indirectly affects N_2O production through its influence on soil water evaporation rate, and subsequently on WFPS. In accordance, the studies reviewed here suggest that this factor, along with water and N inputs, was responsible for the large differences in N_2O emissions that were observed between winter-cropped (usually rainfed) and summer-cropped (irrigated) Mediterranean soils. A positive, significant correlation between temperature and N_2O flux was generally identified (Heller et al., 2010, Lee et al., 2009, Meijide et al., 2007, Petersen et al., 2006, Vallejo et al., 2006). In some cases, such a relationship was not evident throughout the whole experimental period, but only under certain circumstances. For example, low temperatures (below 10-12ºC) were associated with very low (Lee et al., 2009) or even negative (Meijide et al., 2009, Sánchez-Martín et al., 2010b) N_2O fluxes. In the study by López-Fernández et al. (2007), temperature only influenced N_2O flux during the irrigation period; this was probably due to restrictions on N_2O production imposed by low water availability in the pre-irrigation period. Sánchez-Martín et al. (2010a) observed rapid soil desiccation after slurry application due to high temperatures, which delayed nitrification until the onset of irrigation. Other studies found no correlation between N_2O fluxes and temperature during the experimental period (e.g. Sánchez-Martín et al., 2008a, Steenwerth and Belina, 2008). In other climates, Horvath et al. (2010) found a link between N_2O emissions and soil temperatures up to 20°C. Schaufler et al. (2010) reported a non-linear increase in N_2O emissions with temperature for a large set of soil cores studied under laboratory conditions.

pH

The magnitude and origin of N_2O emissions can be markedly affected by soil pH. At low pH, N_2O reduction is usually inhibited, which results in increased emissions (Baggs et al., 2010, Sánchez-Martín et al., 2008a), although this effect may only be apparent in the long term (Baggs et al., 2010). In the experiments reviewed, the soils had pH values ranging from 5.2 to 8.1 (with an average of 7.7 for all the treatments). Only two trials were performed in slightly acid soils: one in Australia (pH=6, Barton et al., 2008, 2010, 2011) and one in California (pH=5.2-7.2, Garland et al., 2011). The N_2O emissions in those trials were lower than average, but they all related to extensive systems with low or zero water and N inputs.

Texture

Soil texture affects soil N_2O production through its influence on soil aeration which, in turn, conditions nitrification and denitrification processes. For sediment, denitrification was shown to increase with the proportion of fine-textured sediment, < 50 µm (Garnier et al., 2009). In this study, most of the soils analyzed were medium-textured, with only one study site corresponding to a fine-textured soil (Garland et al., 2011) and two corresponding to light-textured soils, one of which was in Australia (Barton et al., 2008, 2010, 2011) while the other was in California (Steenwerth and Belina, 2008). N_2O emissions were lower than average in the three mentioned sites, but the differences cannot be attributed to soil texture due to the low number of studies.

3.3 The influence of fertilization type and irrigation management on biochemical processes driving the annual pattern of N_2O emissions

The production of N_2O, whether in soils, wastewater treatment plants, sediments or water bodies, mainly results from biological transformations of nitrogenous compounds (Wrage et al., 2001). As Firestone & Davidson (1989) reported, soil aerobic microorganisms can nitrify soil NH_4^+ to NO_3^-, with N_2O being emitted as a by-product of this transformation. This NO_3^- can then be sequentially reduced to nitric oxide (NO), N_2O and finally N_2 by anaerobic denitrifiers; N_2O emissions occur when the reduction is incomplete. Finally, nitrifier denitrification has been proposed as a third pathway for N_2O production in the soil (Kool et al., 2011, Wrage et al., 2001). Through this pathway, autotrophic nitrifiers oxidize NH_3 to nitrite (NO_2^-) and then reduce NO_2^- to NO, N_2O and N_2 (Wrage et al., 2001).

In real agroecosystems, there is rapid shifting between the different pathways in line with changes in environmental factors and management activities. The typical Mediterranean climate pattern includes a very marked drought period during summer and usually has mild temperatures and an erratic distribution of rainfall over the rest of the year; this contributes to the existence of several wetting and drying cycles. When the soil is rewetted, without reaching complete anoxia, microbial activity is recovered, leading to a pronounced peak in N_2O and CO_2 emissions; this is what has been called the "pulse" or "Birch" effect (Birch, 1958, Beare et al., 2009, Davidson et al., 1993). This suboptimal activity is enhanced by the availability of large quantities of C and N substrates that have been accumulated in the soil due to the death of soil microorganisms during the previous dry period. A large proportion of the annual N_2O fluxes in Mediterranean cropping systems is comprised of pulses that occur after rainfall or, when present, irrigation events, especially when the soil was previously dry. N_2O pulses after rainfall events are also common in temperate climates, where they may be driven by denitrification (Davidson et al., 1993; Vilain et al., 2010). Pulses may also be driven by aerobic processes (nitrification) when the climate conditions are semi-arid (Galbally et al., 2008). Pulses driven by nitrification are, however, usually of lower intensity, in absolute terms, than those driven by denitrification (see section 3.3.2).

On the other hand, all of these biochemical processes can also occur in the soil simultaneously due to the high complexity of soil structure, where adjacent microsites can show very different levels for such soil parameters as WFPS, NH_4^+,

NO_3^- or C accumulation. This spatial and temporal heterogeneity has already been specifically studied in Mediterranean cropping systems. Kong et al. (2011) studied the abundance of ammonia oxidizing bacteria (AOB), denitrifiers and total bacterial communities in different soil microenvironments under different long-term management regimes. They showed that despite the fact that AOB and denitrifier community abundances were affected by management practices, they were largely decoupled from N cycling. Nitrifier and denitrifier communities were larger and fluctuated more in microaggregates than in particulate organic matter (POM) and silt-and-clay fractions. These findings suggest that microaggregates are potential hotspots for N_2O production in the soil(Kong et al., 2011).

Although nitrification and denitrification are the processes responsible for N_2O production in the soil, they may not be correlated with N_2O flux. Denitrification activity could actually shift from being a source to a sink for N_2O, depending on its efficiency for N_2O reduction to N_2. This efficiency has sometimes been shown to be enhanced under Mediterranean conditions, thus potentially decreasing N_2O emissions. This process may occur at relatively low WFPS (Menéndez et al., 2008), but it is usually enhanced when N_2O diffusivity is low due to very high WFPS (Lee et al., 2009, Sánchez-Martín et al., 2010b, Vallejo et al., 2005) and when soil NO_3^- content is low (Ryden and Lund, 1980, Sánchez-Martín et al., 2008a, 2010b). Labile organic C sources could also increase denitrification efficiency (section 3.3.1). Whether driven by this or by another process, a net N_2O uptake has been observed on some occasions (Barton et al., 2008, 2011, Garland et al., 2011, Meijide et al., 2009, Sánchez-Martín et al., 2010b). N_2O uptake events occurred at the end of spring, coinciding with low WFPS and low mineral N levels and/or high DOC (Barton et al. 2008, Garland et al., 2011). The information related to the different biochemical pathways for N_2O production that were influenced by seasonal changes in environmental factors is reviewed in the following sections and grouped according to fertilizer type and water management regime.

3.3.1 Fertilizer type

Organic fertilizers are very heterogeneous materials, whose properties can vary widely depending on their origin and processing. As a general trend, however, adding organic matter to the soil provides the labile C substrates needed for denitrification, which is further enhanced by the creation of anaerobic microsites, even when soil WFPS is <55% (García-Ruiz and Baggs, 2007). The positive effect of a range of different organic fertilizers on the denitrification rate has been verified by López-Fernández et al. (2007), Meijide et al. (2007) and Vallejo et al. (2006), in irrigated arable plots in Central Spain, in which a larger proportion of N_2O fluxes were driven by denitrification in soils amended with organic matter than in synthetic N fertilized soils. On the contrary, in drip-irrigated treatments (Sánchez-Martín et al., 2008b), digested NH_4^+-enriched pig slurry-amended plots produced proportionally less N_2O by denitrification (56%) than the unfertilized control (92%).

DOC (see section 3.2.3) is a soil parameter which is affected by the type of fertilizer employed. Applying mineral N promotes the consumption of DOC by soil microbial biomass (Sánchez-Martín et al., 2008b) and applying urea is usually related to a temporal (1-2 months) increase in DOC (Meijide et al., 2007, Vallejo et al., 2006). Complex organic materials like composts release labile C compounds during their mineralization process, even though their DOC promoting effect could be delayed until 3-4 months after application (Meijide et al., 2007). DOC concentrations have been correlated with the denitrification rate in numerous cases (López-Fernández et al., 2007, Meijide et al., 2007, Vallejo et al., 2006) and especially during the irrigation period. High DOC during irrigation promoted anoxia, which favored denitrification, but not necessarily N_2O emissions (López-Fernández et al., 2007, Meijide et al., 2007). This implies that labile C could also reduce the N_2O/N_2 ratio. Indeed, enhanced denitrification efficiency due to the availability of labile C resulted in a reduction in net N_2O emissions in a Cumulic Haploxeroll in California (Steenwerth and Belina, 2010).

A second possible explanation for N_2O reduction by organic fertilizers was pointed to by Dick et al (2008) for semi-arid cropping systems in Mali, where lower N_2O emissions were measured in plots receiving mixtures of urea and organic manure as opposed to those that only received urea. These authors suggested that N could be immobilized more efficiently by the existing microbial biomass when easily available C and N are simultaneously added to a soil lacking C and N. For example, adding different organic residues to Mediterranean soils fertilized with synthetic N reduced denitrification losses as a percentage of applied N (Coskan et al., 2002), which could perhaps be explained by more efficient N immobilization. However, this immobilization would not necessarily explain the results obtained in Mediterranean systems, where denitrification is usually promoted by organic fertilizers.

The nitrifier denitrification pathway is favored at low concentrations of available C and sub-anoxia (Wrage et al., 2001, see next section); these are common characteristics of Mediterranean soils. The addition of organic C sources to the soil could therefore influence this pathway. An increase in C availability could reduce the contribution of this pathway to overall N_2O emissions, as compared to that of other pathways, and this could help to explain the reduction in cumulative N_2O emissions observed in our analyses.

3.3.2 Rainfed systems

Low N_2O emission levels in Mediterranean rainfed cropping systems are conditioned by the typical agro-climatic features of these systems. During the winter season, N_2O production is usually limited by temperature but also by other factors such as low levels of WFPS (Barton et al., 2008, 2010), soil organic matter (Sánchez-Martín et al., 2008a) or mineral N (Lee et al., 2009). A strong coupling between N mineralization and immobilization has been reported at low soil temperatures (Barton et al., 2010). Moreover, the N fertilization rate in rainfed systems under these climatic conditions is generally low (Kroeze et al., 1998, Ryan et al., 2009) due to crop growth limitations driven by climate. This practice could also contribute to low N_2O emissions.

In late spring, during maturation of winter crops, high temperatures favor microbial processes, but N_2O production may be limited by low NH_4^+ content and low WFPS (Meijide et al., 2009).

During summer, dry soil conditions usually prevent N_2O emissions, except when significant rainfall occurs (Barton et al., 2008, 2010, 2011). Rainfall events after the summer are usually related to significant N_2O pulses due to N mineralization and the subsequent accumulation of mineral N in the soil during this period (Meijide et al., 2009). This seasonal pattern of soil N dynamics also occurs in Mediterranean natural ecosystems (Ochoa-Hueso and Manrique, 2011).

In Mediterranean rainfed systems, and especially in those cultivated in well-aerated soils, low rainfall leads to low WFPS and therefore to a high redox potential which is unsuitable for denitrification (Lugato et al., 2010). According to most of the studies reviewed, this makes nitrification the most usual pathway for N_2O production in low organic matter, non-irrigated Mediterranean soils (Barton et al. 2008, 2011, Lugato et al., 2010, Meijide et al., 2009, Menéndez et al., 2008). This hypothesis is also supported by the positive relationship between N_2O fluxes and soil NH_4^+ levels (Meijide et al., 2009). N_2O pulses due to nitrification are relatively small (Sánchez-Martín et al., 2010a), which is in line with the typically low cumulative N_2O fluxes found in Mediterranean rainfed systems.

N_2O emissions due to denitrification can be common in rainfed systems after heavy rainfalls or when the rainy season is especially wet (Meijde et al., 2009, Sánchez-

Martín et al., 2010b). Complete anoxia promoting the reduction of N_2O to N_2 is unlikely or very transient in Mediterranean rainfed systems. Therefore, wetter-than-average cropping years usually lead to higher N_2O emissions than drier ones (Sánchez-Martín et al., 2010b), which may also happen under temperate climatic conditions (Laville et al., 2011).

Kool et al. (2011) showed that the nitrifier denitrification pathway for N_2O production could be responsible for a significant fraction of N_2O emissions in agricultural soils, especially under moisture conditions that are sub-optimal for denitrification. Following this logic, this pathway could be important in N_2O production under Mediterranean conditions (Mondini et al., 2007, Sánchez-Martín et al., 2008a), particularly in rainfed and drip-irrigated systems, where the moisture content required for denitrification is not often reached. For example, Sánchez-Martín et al. (2008a) hypothesized that most N_2O was produced by nitrifier denitrification in a Mediterranean soil incubated under laboratory conditions at 40% WFPS and identified it as a significant source at 90% WFPS. Nitrifier denitrification has also been proposed as a possible pathway for N_2O uptake when conditions are not suitable for anaerobic denitrification (Chapuis-Lardy et al., 2007, Meijide et al., 2009). If nitrifier denitrification plays an important role in N_2O production and consumption in Mediterranean soils, this would imply that a significant fraction of N_2O emissions currently attributed to denitrification or nitrification could actually be driven by this biochemical pathway. Such uncertainty points to the need for more research in order to accurately understand the biochemical processes underlying N_2O emissions in Mediterranean agroecosystems.

3.3.3 Irrigated systems

Irrigation is associated to the intensification in N inputs, and in Mediterranean cropping systems it usually takes place in late spring and summer, when temperatures are highest. Therefore, the most relevant physical properties of the soil (temperature and water and N availability) are optimal for N_2O production during the cropping season. During the winter fallow period, the conditions are similar to those described for rainfed systems, though more residual N can be available in the soil due to the higher application rates of N fertilizers.In summer-irrigated Mediterranean systems, it is therefore usual for significant N_2O fluxes to occur throughout the annual cycle.

In the case of high-water irrigation techniques, such as furrow irrigation, near-saturation conditions are transiently reached for one or more days after irrigation. These conditions usually lead to a very high initial N_2O pulse after the first irrigation event (López-Fernández et al., 2007, Sanchez-Martin et al., 2008b, 2010a, Vallejo et al., 2005). This can then be followed by two or more large pulses in the course of the irrigation period (Kallenbach et al., 2010, Vallejo et al., 2006). Denitrification is usually the prevailing N_2O production pathway under this type of irrigation (Sánchez-Martín et al., 2008b); this is usually a minor pathway before the onset of irrigation and then becomes the main source during the irrigation period (López-Fernández et al., 2007, Vallejo et al., 2005), accounting for up to 99% of N_2O fluxes (Sánchez-Martín et al., 2010a). Nitrification is likely to occur in these systems when a high concentration of NH_4^+ is reached in the soil; this is sometimes the case when synthetic or liquid organic fertilizers are applied (Meijide et al., 2007, Sánchez-Martín et al., 2010b, 2010c, Vallejo et al., 2006), providing aerobic conditions are met, which can be the case in high-water irrigated systems, where WFPS fluctuations are very large.

N_2O fluxes are generally lower in drip-irrigated soils than in those receiving high water applications, and the emission patterns also differ. Drip irrigation promotes a small but steady flux of N_2O throughout the cropping season (Kallenbach et al., 2010), which could be accompanied by small N_2O pulses after each irrigation event (Sánchez-Martín et al., 2008b, 2010a) instead of one or various large pulses of the sort typically associated with furrow irrigation systems. As in rainfed systems, low water availability in drip-irrigated soils results in nitrification becoming the most important source of N_2O (Kallenbach et al., 2010). This assumption is supported by papers that report a lack of relationship between N_2O fluxes and NO_3^- concentrations (Garland et al., 2011, Steenwerth and Belina, 2008) and large increases in the size of the soil NO_3^- pool after the NH_4^+ peak (Sánchez-Martín et al., 2008b). Indeed, N_2O production by denitrification is prevented by low water availability, as the soil rarely exceeds 60 % WFPS, either with subsurface (Kallenbach et al., 2010) or surface (Sanchez-Martin et al., 2008b, 2010a) drip irrigation techniques. However, according to the latter works, the spatial distribution of soil humidity, and subsequently of N_2O fluxes, stress the need for a stratified sampling of N_2O fluxes and soil parameters. For example, wet areas near drippers may lead to a N_2O source, but a WFPS of >80% can be locally reached in dripping points, which may promote denitrification to N_2.

4. Indirect sources of N_2O emissions4.1 Upstream emissions

Emissions during the production, manufacturing and transport of fertilizers play a key role in total fertilizer emissions. It is not reasonable to consider these processes from a specifically Mediterranean perspective given the wide range of conditions under which fertilizers are produced in these areas. Nonetheless, there are general differences in the emission of GHG during the production of synthetic and organic fertilizers that are worth noting.

Synthetic fertilizers

The production of synthetic fertilizers requires a high consumption of fossil energy to reduce N_2 to NH_3. According to the IPCC (2006b), average CO_2 emissions due to NH_3 production in European plants range between 2.55 and 3.57 kg CO_2per kg of fixed N, depending on the technology employed. In comparison, N_2O emissions from the soil calculated using IPCC EF are equivalent to 4.68 kg CO_2per kg of applied N. In Mediterranean rainfed and drip-irrigated cropping systems, where N_2O EF is lower than the IPCC default EF, these pre-farm GHG emissions related to fertilizer production may actually be much greater than the on-farm N_2O emissions. For example, Biswas et al. (2008) performed a life cycle assessment (LCA) of rainfed wheat production in the Mediterranean-climate region of Western Australia. They estimated that to produce one ton of wheat, 103.87 kg CO_2-eq was emitted as a result of urea production, whereas N_2O emissions from the field represented 26.98 or 175 kg CO_2-eq, according to whether region-specific (Barton et al., 2008) o IPCC (2006b) N_2O EF was employed.

Organic fertilizers

The use of organic materials as fertilizers requires the management of organic wastes. When the residual organic matter is not produced in the field, it needs to be handled, stored, transported, and sometimes transformed into more stable and easier to handle compounds. This management process is associated with GHG emissions. In the EU-27, N_2O emissions during the housing and storage of animal manure are estimated to be only slightly lower than those associated with their land application (Oenema et al., 2009). GHG emissions related to the production of organic fertilizers from organic waste should be accounted for by comparison with the emissions associated with conventional residue management (e.g., Kim and Kim, 2010, Prapaspongsa et al., 2010). A careful and site-specific assessment of GHG emissions is required during waste management in order to quantify upstream GHG emissions by organic fertilizers.

Legumes are virtually the only organic source of newly fixed N. N_2O emissions during N fixation by legumes are generally taken to be zero or negligible (IPCC, 2006a) and this has been verified under Mediterranean conditions (Barton et al., 2011). Nonetheless, N_2O emissions that occur during legume crop growth and indirect GHG emissions related to their cultivation should be taken into account in full GHG comparisons when legumes are used as green manure. Furthermore, land occupation associated with biological N fixation calls into question the possibility of a considerable substitution of Haber-Bosch-produced N. Indeed, some authors have shown that the internalization of energy and nutrient fluxes in sustainable agriculture may also have a "land cost" that is externalized in fossil fuel-based systems (Guzmán Casado and González de Molina, 2009). Even so, there is still great potential for reducing the N surplus and increasing N fixation in Mediterranean agroecosystems without needing to occupy any extra land (section 5), even if the extent to which sources of organic N can replace synthetic ones still remains unclear.

Transport

The use of organic fertilizers (i.e. slurries, manures) requires large amounts of energy due to their weight. However, their production sources tend to be more local to the end user than synthetic fertilizers, which are generally produced in a few large manufacturing plants. Even without taking into account transport costs for synthetic fertilizers, Wiens et al. (2008) estimated that the distance that liquid pig manure was transported could be increased to 8.4 and 12.3 km, respectively, before the energy cost per kg of available N associated with this manure was equivalent to that of anhydrous ammonia or urea N. Nearby land could therefore receive the resulting manure at an appropriate rate and without high transport costs, as long as the concentration of livestock is not very high (section 5.3).

4.2 Downstream emissions

NO_3^- leaching and NH_3 and oxidized N compounds (NO_x) volatilization are considered the main processes responsible for fertilizer-associated N_2O emissions outside the cropping system and are the only ones classified as "indirect fertilizer N_2O emissions" in the IPCC guidelines for GHG inventories (IPCC, 2006a). Downstream indirect emissions were estimated to represent about 13-17 % of direct emissions in one temperate river basin (Garnier et al., 2009).

NO_3^- leaching

This is a source of major concern in many areas in Mediterranean countries, such as Spain (Lassaletta et al., 2009, 2010, Peña-Haro et al., 2010), because of its eutrophication potential and the negative impact on drinking water from surface or ground waters. In Mediterranean cropping systems, NO_3^- leaching can occur either in irrigated fields (Allaire-Leung et al., 2001) or be related to rainfall events during the rainy season (Angás et al., 2006), and it can be responsible for the loss of up to 25 % of applied synthetic N in a normal winter fallow period (Sánchez-Martín et al., 2010b).

Very large N surpluses can occur from organically fertilized soils, resulting in NO_3^- leaching and related aquifer pollution. For example, the application of high rates of slurries has caused high N losses, in the form of NO_3^-, in intensive livestock production areas in NE Spain (Peñuelas et al., 2010). Even so, when organic and synthetic fertilizers are compared at similar N application rates, NO_3^- leaching is generally significantly lower with organic fertilizers (Antoniadis et al., 2010, Díez et al., 1997, 2000, Celik, 2009, Sánchez-Martín et al., 2010a). Some authors have even found N leaching to be lower in soils fertilized with compost than in unfertilized plots (Tejada and González, 2006). In general, low (Sánchez-Martín et al., 2010a) or null (Díez et al., 2004) NO_3^- leaching reductions are associated with liquid, highly-mineralized organic fertilizers, such as pig slurries. On the other hand, other approaches to nutrient management based on organic matter cycling, such as cover cropping, have also proved capable of strongly reducing NO_3^- leaching in Mediterranean environments (Salmerón et al., 2010, Steenwerth and Belina, 2008, Wyland et al., 1996).

The lower N leaching associated with organic amendments could be driven by a decrease in soluble N in the soil due to the increased performance and efficiency of denitrifiers (Kramer et al., 2006, Steenwerth and Belina, 2010), the immobilization of NO_3^- by a larger microbial biomass (Burger and Jackson, 2003), or the capture of N in the SOM which is built up by the addition of organic matter. For example, Kong et al. (2007) reported that an additional 590 kg N ha^{-1} had been stored in the soil after 11 years of organic management, along with the sequestration of 5.7 Mg C. Applying organic matter to the soil could also reduce the subsoil NO_3^- pool by enhancing the activity of denitrifying microorganisms in the subsoil or groundwater (Sánchez-Martín et al., 2010a), as organic C is the main limiting factor for denitrification in the subsoil (Haag and Kaupenjohann, 2001). In an experiment performed under Mediterranean conditions, DOC leaching of 2.3-4.8 kg C ha^{-1} helped to complete the reduction of 2.1-4.5 kg NO_3^--N ha^{-1} to N_2 (Sánchez-Martín et al., 2010a), implying capacities for subsoil NO_3^- removal of 100, 13.2, 10.4 and 6.7 % for organic manure, control, digested pig slurry and urea treatments, respectively. These results should, however, be interpreted with care since incomplete subsoil denitrification could also release N_2O, which could be transported by drainage water due to its high solubility (Van Cleemput, 1998).

NH_3 Volatilization

We did not find any field studies that compared NH_3 volatilization after organic and synthetic fertilization under Mediterranean climatic conditions, although some separate data are available. Various different synthetic fertilizers applied to wheat under simulated Mediterranean conditions were reported to have released 12-38 % of their N (mostly as NH_3) (Buresh et al., 1990), whereas at arid and semi-arid sites in Syria, slightly lower gaseous N losses, of 11-18% (again mostly as NH_3), were recorded after urea application (Abdel-Monem et al., 2010). The only micrometeorological studies conducted that specifically measured NH_3 losses reported: i) 10.1 % NH_3-N losses from urea (Sanz-Cobena et al., 2008); ii) 5 % losses after green manuring (Rana and Mastrorilli, 1998) and iii) 20 % of the Total Ammonium Nitrogen applied with a pig slurry spread at the soil surface (Sanz et al., 2010). The first two values, for synthetic and organic fertilization, are below the 20 % default EF value established by the CORINAIR Emission Inventory Guidebook for regions with spring temperatures >13.8º C (CORINAIR, 2006). Contrastingly, measured NH_3 losses, from synthetic and liquid organic manures, will be in accordance with the values proposed by the IPCC (10 and 20 % for synthetic and organic fertilizers, respectively). Existing discrepancies could be associated with local climatic, soil and management conditions, e.g. dry conditions during the experimental period, the presence of vermiculites as the main clay mineral, and the application of 10 mm of irrigation immediately after fertilizing, all possibly favor large decreases in the availability of exchangeable NH_4^+, which can be potentially lost as NH_3 (Sanz-Cobena et al., 2008).

N_2O emissions

There is very little information about which fraction of the N lost from the cropping system in the form of NO_3^-, NH_4^+ or NH_3 is finally transformed into N_2O in Mediterranean environments. The available data suggest that this fraction could be very significant but variable. Measurements performed in the Douro Estuary in Portugal (Teixeira et al., 2010) revealed that 0.5-47 % of the N gases produced were in the form of N_2O, and that emissions were correlated with sediment organic matter. On the plain of the River Po, in Northern Italy, springs were found to be supersaturated with N_2O and were subject to a significant degassing process. As a result this area had a very high potential as a source of N_2O and other GHG gases (Laini et al., 2011). In one stream in the Doñana National Park, SW Spain, NO_3^- pollution which originated in nearby agricultural fields, mostly from synthetic N sources, was associated with N_2O production, but also with that of CH_4 and CO_2 (Tortosa et al., 2011).

5. Mitigation options with organic fertilizers

5.1 Water-saving agricultural systems

The significant reduction in N_2O fluxes in rain-fed and drip irrigated systems as opposed to conventionally irrigated systems (sections 3.2.1, 3.3) implies a high potential to mitigate N_2O emissions through the optimization of water use. However, the reduction in N_2O fluxes achieved by applying drip irrigation may only occur when a source of organic matter is applied (Kallenbach et al., 2010). Drip irrigated systems foster water and N_2O emission savings while maintaining yields (Tognetti et al., 2003, Kallenbach et al., 2010), whereas in rainfed systems, yield-scaled emissions maybe affected by a lower productivity (section 6.3). However, a full GHG accounting should also consider the higher fossil energy consumption in irrigated systems (Alonso and Guzmán, 2010).

5.2 Minimization of bare soil

Bare fallows in herbaceous crop rotations and bare soils in woody perennial systems are usually maintained in Mediterranean environments in order to increase water and nutrient availability for commercial crops. This assumption has been challenged by research data, which show that bare fallows may not contribute to overall productivity as much as legume cover crops (López-Bellido et al., 2000, Martín-Rueda et al., 2007), as has also been reported in other dry environments (Rinnofner et al., 2008). Bare fallows and other bare soils may therefore represent stages, or areas, of the cropping systems capable of releasing large quantities of reactive N compounds (which are responsible for both direct and indirect N_2O emissions) without contributing to overall productivity.

Fallow and bare soil emissions can be avoided by system intensification, in which cash crops substitute bare fallows, and also through the cultivation of cover crops, either in crop rotations or in perennial systems. Cover crops have a large potential for increasing N retention in cropping systems and thereby reducing indirect N_2O emissions, mainly through i)N immobilization by catch crops (e.g., McSwiney et al., 2010, Gabriel and Quemada, 2011), ii) biological N fixation with legume green manures (e.g. Rinnofner et al. 2008) and iii) soil protection against erosion (Boellstorff and Benito, 2005, Gómez et al., 2009). Their effect on direct N_2O emissions may vary according to the specific case. Legume cropping in the

Mediterranean semi-arid environment of Western Australia yielded similar emission rates as bare soils (Barton et al., 2011). In California, Kallenbach et al. (2010) recorded higher N_2O emissions from a legume cover cropped treatment than from bare soil. These authors suggested that non-legume cover crops could help to reduce N_2O emissions due to their higher C:N ratio and deeper roots, which could extract soil N more efficiently. However, in order to maintain the benefits of biological N fixation, we would propose trials with mixtures of legumes and non-legumes and also their combination with low-quality organic residues (section 5.4).

5.3 Improved waste management

The high population densities in most Mediterranean areas suggest that urban wastes could represent a significant source of organic matter for agricultural fields. Municipal solid waste and sewage sludge, especially if composted, usually show good agronomic performance, and they also promote an increase in soil organic carbon (Diacono and Montemurro, 2010). As we have already seen in this review, the use of these materials as fertilizers can also help to reduce direct and indirect N_2O emissions. In spite of these advantages, heavy metals and other toxic compounds may call into question their safe application to soils, which points to the need to appropriately separate urban organic wastes at source.

In the case of livestock farming, the continued specialization of livestock production units leads to increasing problems with the safe recycling of manure nutrients (Petersen et al., 2007), and encourages their application to nearby soils in high doses, which boosts N losses, and particularly N_2O ones. In some parts of Israel, for example, organic manures are applied at rates of over 1000 kg N ha^{-1}, which increases emissions to 34.4 kg N_2O-N ha^{-1} (Heller et al., 2010). The minimization of N surpluses that cause both pollution and dependence can be achieved by the circulation of materials between livestock and cropping systems, as demonstrated in other regions (Nekomoto et al., 2006). Biogas production is another integrated approach to waste management that can help to reduce GHG emissions.

5.4 Tightening the N cycle through N immobilization

Typical woody crops cultivated under Mediterranean climatic conditions include vines, olives, almonds, walnuts, citrus and other fruit trees. They occupy large areas and produce high quantities of pruning residues, which are normally burned in the field, resulting in emissions of trace GHG and also stored C. Proposals have recently been made for their use as energy sources (e.g. Di Giacomo and Taglieri, 2009, Kroodsma and Field, 2006). An alternative, or complementary option, is to incorporate these residues into the soil; this could greatly enhance soil carbon storage and biodiversity (Holtz and Caesar-Ton That, 2004) and would also protect the soil against erosion (Rodríguez Lizana et al., 2008), while fostering a reduction in N losses. Indeed, N retention through N immobilization during fallow periods cannot only be achieved with catch crops (section 5.2), but also through the addition of low quality organic residues (Muhammad et al., 2011, Sakala et al., 2000). To be more specific, lignin and polyphenol rich materials, such as pruning residues, have been associated with reductions in N_2O fluxes over a broad range of conditions due to their strong N immobilization effect (Frimpong and Baggs, 2010, García et al., 1997, Gomes et al., 2009). In a laboratory experiment, García-Ruiz and Baggs (2007) demonstrated that N fertilizer application did not increase N_2O emissions if the soil had been mixed with olive leaves; this was related to the high lignin (11%) and polyphenol (2%) contents of olive residues.

The major concern here, is that N immobilization may negatively affect crop production (Frimpong and Baggs, 2010, Soumare et al., 2002), especially during the first months after application (Soumare et al., 2002). Potential reductions in crop productivity due to N immobilization could, however, be remedied by appropriately combining and timing the application of N sources with different mineralization kinetics. In this sense, mineralization kinetic parameters could be used to evaluate the most suitable N release pattern for organic fertilizers (Marinari et al., 2010). Successful examples of soil protecting practices in three different Mediterranean orchards, including the addition of pruning residues, led to a significant increase in fruit yield compared with conventional management strategies (Montanaro et al., 2009, Sofo et al., 2010).

6. Information gaps

We found a number of knowledge gaps that we would recommend addressing in future field research.

6.1 Length of the experiment

The studies included in this review were carried out on average for 243 days. The average experiment length was consistently higher in rainfed systems than in irrigated ones, where cropping period is usually shorter. Measurement periods of less than 1 year do not account for total annual emissions, e.g. those of the residual effects of fertilizers, which could lead to a possible underestimation of EF. An estimation of yearly cumulative emissions and EF based on simple modeling of measured emission levels resulted in a great increase in emissions in irrigated groups (Tables 2 and 3). This procedure, however, could overestimate yearly emissions because N_2O flux is usually highest during cropping period, when fertilizers and water are applied.

Significant emissions during the post-harvest fallow period have been recorded in Mediterranean cropping systems. In the summer fallow period of rainfed systems, emissions are relatively low, most of the time, due to the dry soil conditions (Steenwerth and Belina, 2008), and isolated summer rainfall only stimulates N_2O emissions very transiently due to rapid soil desiccation (Sánchez-Martín et al., 2010a). Large or multiple rainfall events during summer can, however, increase summer fallow emissions to 55% of yearly emissions (Barton et al., 2008, 2010, 2011). Significant quantities of N_2O may also be lost during the winter fallow periods of summer-cropped, irrigated fields (Burger et al., 2005, Kallenbach et al., 2010, Lee et al., 2009, Sánchez-Martín et al., 2010a).

As previously mentioned, there is particular concern about the residual effect of organic fertilizers, given their typically slow and extended N release, which can prolong their N_2O emission period (Jones et al., 2007) and would justify the use of “available N” rather than “total N” when calculating their EF (e.g., Vallejo et al., 2006, see section 2 of this paper). For example, Meijide et al. (2009) found various N_2O emission peaks during the post-harvest period in organic-fertilized soils as opposed to only one peak in the control and synthetic treatments. Conversely, releasing labile C compounds during the mineralization of organic fertilizers in soils poor in organic matter could help to prevent N_2O emissions (Sánchez-Martín et al., 2010a). These findings challenge the assumption that N_2O fluxes would tend to be extended by the residual effect of organic fertilizers and underline the need for longer sampling periods and more long-term studies.

Moving beyond short term residual effect, Li et al. (2005) argued that techniques that promote C sequestration could enhance N_2O emissions in the long term due to the increase in soil organic carbon (SOC). Under Mediterranean conditions, long term experiments comparing organic and synthetic fertilization have only been performed at one site: Russell Ranch in Davis, California (Burger et al., 2005, Kong et al. 2007, 2009). None of these studies reported increases in N_2O emissions, despite the fact that SOC pool increased significantly after up to 11 years of organic management. We therefore hypothesize that under Mediterranean conditions there may be a SOC content threshold above which fertilizer-related N_2O emissions would start to increase. In spite of this, at relatively low SOC levels around 1 %, such as those studied by Kong et al. (2007, 2009), organic fertilizers would help to reduce N_2O fluxes.

6.2 Background emissions

Unfertilized control treatment N_2O emissions on average represented 63.5 % of fertilized treatment emissions in the reviewed studies. This implies that a very large fraction of soil N_2O emissions cannot be explained by the fertilizer EF approach. The pulsing effect probably contributes to these high background emissions. Changes in management practices, such as irrigation methods, may also induce distinct changes in emissions from control and fertilized treatments, which would also affect N_2O EF. For example, Sánchez-Martín et al. (2010b) reported a sharp reduction in N_2O emissions with drip irrigation, as opposed to furrow irrigation. The reduction was, however, much greater in the control treatment; it produced greater EF values for drip-irrigated treatments, while cumulative emissions were actually lower than for furrow-irrigated treatments.

These data suggest that background emissions are influenced by management and consequently require specific accounting, as they are susceptible to improvement. The calculation of EF, by dividing cumulative N_2O emissions by the applied N rate, without subtracting unfertilized control emissions (e.g. Dobbie and Smith, 1999), is an approach that includes background emissions in estimations based on applied fertilizer. This method would be less useful, however, if these background emissions were affected by other management operations. We therefore recommend complementing EF data with cumulative emission data for every treatment, including unfertilized controls, when presenting research results.

6.3 N_2O EF and yield-scaled EF

EF was only provided in 52 % of the reviewed studies that measured field N_2O emissions. In the other cases, it was not possible to calculate EF because unfertilized treatments were absent.Despite its limitations, EF provides a useful simplified tool for upscaling the emissions of a given region based on fertilization rates, provided that background emissions are also accounted for. Although N_2O emissions vary greatly, the EF approach is fairly well supported by field data (Stehfest and Bouwman, 2006, Petersen et al., 2006). However, the results analyzed in this review suggest that it needs region-specific modulations for such factors as fertilizer type and irrigation type.

Yield-scaled EF is also a very informative parameter for understanding site-specific trade-offs between fertilization type, N_2O fluxes and yield performance. Full GHG accounting methodologies such as LCA could greatly benefit from this information, as they are usually product based. When applied to a single type of fertilizer, yield-scaled EF usually reveals that N_2O emissions are smaller at intermediate N application rates (Hoben et al., 2011, Van Groenigen et al., 2010). When comparing management schemes, yield-scaled EF shows that environmental benefits may disappear if they are associated to lower yields (de Backer et al., 2009). In the Mediterranean context, Meijide et al. (2009) found that the performance of organic and synthetic fertilizers slightly differed according to whether they were evaluated on an applied N basis or on a yield basis. The authors also warned about the annual variability of crop yield, which should be taken into account when applying yield-scaled index.

Other studies comparing yields associated with organic and synthetic fertilization in Mediterranean cropping systems have obtained heterogeneous results, although most of the authors consulted reported similar yields for the two types of fertilization (Altieri and Esposito, 2008, Bilalis et al., 2010a, 2010b, Caporali and Onnis, 1992, Clark et al., 1999, Díez et al., 1997, 2000, Deria et al., 2003, Drinkwater et al., 1995, Efthimiadou et al., 2009, Herencia et al., 2007, Lithourgidis et al., 2007, Madejón et al., 2001, Meijide et al., 2007, Montanaro et al., 2009, Montemurro et al., 2005, 2008, 2010, Morra et al., 2010, Pardo et al., 2009, Vallejo et al., 2006). Higher yields for organic fertilization have also been reported (Campiglia et al., 2011, Curuk et al., 2004, Deria et al., 2003, Karamanos et al., 2004, Madejón et al., 2001, Melero et al., 2006, Montemurro et al., 2008, Sofo et al., 2010, 2010b), while other authors discovered yield reductions related to organic as opposed synthetic fertilizers (Annicchiarico et al., 2010, Denison et al., 2004, Deria et al., 2003, García-Martín et al., 2007, Kavargiris et al., 2009, Montemurro et al., 2006, 2007, 2009, 2010, Morra et al., 2010). The disparity in the relative yield performance of organic and synthetic fertilization is understandable given the variety of types of organic fertilizers, management techniques and agro-climatic conditions. This complexity further emphasizes the need for yield data in specific N_2O emission studies.

6.4 Cropping systems that require more research

Different crop types have been unevenly studied. Maize, open-air horticultural crops and winter cereals have been studied under a fairly wide range of conditions and we now have a rough picture of the behavior of their N_2O emissions in these systems, but information about other very important Mediterranean crop types is almost nonexistent. Despite the importance of many of the woody perennial crops grown in this biome, vines are the only crop in this category for which N_2O emissions have been measured. Greenhouse horticulture must also be studied, as its particular environmental conditions, including high humidity and temperature throughout the year, would probably affect the pattern of N_2O emissions.

Organic management systems also require specific research, as they not only exclude synthetic fertilization, but also the use of other synthetic compounds which may either increase or reduce N_2O emissions (Kinney et al., 2005, Spokas et al., 2006). Organic farming is a very important management option in Mediterranean areas. For example, the two European countries with the largest surface areas under organic farming are Spain and Italy (Willer and Kilcher, 2011), whereas California is the state with the largest organic acreage in the USA (USDA, 2011). Lower N_2O fluxes have been reported for organic fields under Mediterranean conditions (Burger et al., 2005, Kong et al., 2007, 2009, Petersen et al., 2007), but the available data is very limited and yield-scaled performance may be reduced by lower yields (Kong et al., 2009).

6.5 Full accounting of GHG emissions

Management recommendations based only on direct emissions may not meet abatement objectives if they are based on techniques that increase emissions at any other point in the life cycle of the fertilizers in question. There is a distinct absence of comparisons of full upstream GHG emissions between synthetic and organic fertilizers for Mediterranean cropping systems. Research into downstream emissions has mainly focused on NO_3^- leaching, as NH_3 volatilization has hardly been studied at all. Furthermore, there is very little data on which fraction of this N finally forms N_2O, although the available evidence suggests that it may be very significant (section 4.2). The simultaneous estimation of the emissions of all of the GHG involved in the GWP of cropping systems can be facilitated by modeling approaches. For example, the DAYCENT model has been shown to be capable of accurately predicting soil GHG emissions for a series of Mediterranean cropping systems (De Gryze et al., 2009), and DNDC is another interesting option (Lugato et al., 2010). At the farm or final product level, LCA methodologies should be adjusted to Mediterranean environments using a specific EF (Biswas et al., 2008, 2011).

7. Concluding remarks

The data reviewed suggest that organic fertilizers and water saving techniques could reduce agricultural N_2O emissions under Mediterranean climatic conditions. However, the number of experimental sites at which emissions from organic and synthetic fertilizers have been compared is very limited. In the first part of our analysis, which included the majority of the published data, cumulative N_2O emissions were significantly lower for solid organic fertilizers than for synthetic and liquid organic fertilizers. In a more detailed approach, a meta-analysis of compliant studies revealed that direct N_2O emissions after the application of organic fertilizers were lower than emissions after the addition of synthetic fertilizers (an average 23 % was observed for cumulative emissions and EF). When organic fertilizers were segregated in solid and liquid materials, only solid fertilizer emissions were significantly lower than synthetic. When total N instead of available N was used for EF calculations, solid organic fertilizers achieved a 67 % reduction in EF compared to synthetic fertilizers. A slower mineral N release from organic fertilizers could prevent high soil N levels prone to N_2O losses. Moreover, the differences observed seem to be related to the semi-arid features that are very common in Mediterranean soils, where C and N contents are usually low.

High-water irrigated systems showed the largest losses of fertilizer N as N_2O, whereas these losses were reduced under water-saving irrigation techniques (i.e. surface or subsurface drip irrigation), and were minimal in rainfed systems, in which the N_2O emission response to N fertilizers was reduced by one order of magnitude compared to conventional irrigation and Tier 1 IPCC EF.

Indirect N_2O emissions have not been fully accounted for, but sectorial information suggests a large reduction in N_2O emissions for organic fertilizers. Upstream of the cropping system, substantial reductions in fossil energy and N_2O emissions can be achieved, given that organic fertilizers usually employ waste materials that have not been produced specifically for this purpose. Downstream of the cropping system, Mediterranean data suggest that indirect N_2O emission savings could be achieved by using organic fertilizers on account of their reduced NO_3^- exports.

Options to enhance N_2O mitigation by organic fertilizers include: (i) water management strategies, comprising drip irrigation and rainfed systems; (ii) the minimization of fallow and bare soil emissions through the intensification of crop rotation and the use of cover crops; (iii) improved waste management to reduce indirect emissions, including waste separation at origin, decentralized livestock farming and biogas production; (iv) tightening the N cycle through N immobilization with woody residues in order to minimize emissions during fallow and low crop demand periods.

Identified limitations to the mitigation of N_2O emissions by organic fertilizers include: i) the residual effect as organic fertilizers could prolong the N_2O emission period due to slower N release; ii) the long-term addition of organic fertilizers to soil could enhance N_2O emissions through an increase in SOM content; iii) yield-scaled performance, since organic fertilization could be linked to lower yields, although yield reduction is not the most common response to the substitution of synthetic N sources by organic ones in Mediterranean systems; iv) the availability of total N from organic sources may be constrained by the land cost of legume cultivation. This land cost would be minimized if N fixing crops were cultivated during inter-cropping periods and N losses were reduced, but the extent to which organic fertilizers can replace synthetic ones is so far unknown.

We found a number of knowledge gaps that should be addressed by future field research: i) experiment length is usually <12 months, which can lead to the underestimation of N_2O EF due to failure to account for the residual and long-term effects of fertilizers; ii) background N_2O emissions sometimes represent a large percentage of fertilized treatment emissions; iii) N_2O EF and yield-scaled EF were not always provided in the studies reviewed; iv) some types of cropping system need to be studied in greater depth, including rainfed, drip-irrigated, woody perennial, greenhouse horticulture and organic farming; v) more full cropping system GWP estimations are needed, including indirect emissions and other GHG.

Overall, this review has demonstrated that there is still potential to mitigate N_2O emissions in Mediterranean agriculture through the use of both organic fertilizers and low water management systems. In the first case, the potential lies in comparatively lower N_2O fluxes of organic fertilizers with respect to the use of synthetic fertilizers and in the reduction of indirect emissions both upstream and downstream of the cropping system. In the second, the restriction of water availability that occurs in rainfed and drip-irrigated cropping systems has been shown to effectively reduce N_2O emissions by limiting the microbial processes responsible for N_2O production. Further research is needed to bridge the knowledge gaps in current information and to develop strategies that can fully exploit this N_2O reduction potential without burdening environmental and yield performance.

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4.2. Study 2

Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis.

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Eduardo Aguilera[a, *]; Luis Lassaletta[b, c]; Andreas Gattinger[d]; Benjamín S. Gimeno[e];

[a] *Universidad Pablo de Olavide, Ctra. de Utrera, km. 1, 41013 Sevilla (Spain)*

[b] *UPMC/CNRS, UMR Sisyphe, 4, Place Jussieu, 75005 Paris, France*

[c] *Department of Ecology, Universidad Complutense de Madrid, c/José Antonio Novais s/n 28040 Madrid, Spain*

[d] *Research Institute of Organic Agriculture, Ackerstrasse, CH-5070 Frick, Switzerland.*

[e] *Ecotoxicology of Air Pollution. CIEMAT. Avda. Complutense 22, 28040 Madrid, Spain.*

**Corresponding author: emagufer@upo.es*

Abstract

Mediterranean croplands are seasonally dry agroecosystems with low soil organic carbon (SOC) content and high risk of land degradation and desertification. The increase in SOC is of special interest in these systems, as it can help to build resilience for climate change adaptation while contributing to mitigate global warming through the sequestration of atmospheric carbon (C). We compared SOC change and C sequestration under a number of recommended management practices (RMPs) with neighboring conventional plots under Mediterranean climate (174 data sets from 79 references). The highest response in C sequestration was achieved by those practices applying largest amounts of C inputs (land treatment and organic amendments). Conservation tillage practices (no-tillage and reduced tillage) induced lower effect sizes but significantly promoted C sequestration, whereas no effect and negative net sequestration rates were observed for slurry applications and unfertilized treatments, respectively. Practices combining external organic amendments with cover crops or conservation tillage (combined management practices and organic management) showed very good performance in C sequestration. We studied separately the changes in SOC under organic management, with 80 data sets from 30 references. The results also suggest that the degree of intensification in C input rate is the main driver behind the relative C accumulation in organic treatments. Thus, highest net C sequestration rates were observed in most eco-intensive groups, such as “irrigated”, “horticulture” and controlled experiments (“plot scale”).

1. Introduction

Terrestrial stages of the global carbon (C) cycle are of special importance in the mitigation and adaptation efforts to climate change. Soil organic carbon (SOC) pool is twice as big as atmospheric C pool, and historic losses since 1850 are estimated in 78 ± 12 Pg CO_2 on a global basis, which can be compared to 270 ± 30 Pg CO_2 emitted by fossil fuel combustion (Lal, 2004). Soil-atmosphere net C fluxes are estimated to be low in the present, but there is a large potential to recover the C historically lost, and it has been estimated that 89% of agriculture's greenhouse gas (GHG) mitigation potential relies on C sequestration (Smith et al., 2008). In addition, increasing the SOC content has very relevant benefits for climate change adaptation, because it improves physical, chemical and biological quality of the soil (Lal et al., 2011). These improvements are crucial for sustaining and enhancing crop productivity in a context where climatic conditions become more extreme. On the other hand, many adaptation measures, such as those that reduce soil erosion, conserve soil moisture or diversify crop rotations also promote SOC storage (Smith and Olesen, 2010), which thus appears as a key link between climate change mitigation and adaptation efforts.

Recommended management practices (RMPs) aiming to increase SOC stock must achieve a positive balance between inputs and outputs of carbon through the reduction of SOC losses by oxidation, the increase of organic carbon (OC) inputs to the soil, or a combination of both approaches (Six et al., 2004). Losses can mainly be reduced by minimizing soil disturbance, either with no-tillage or with some other conservation tillage practice, such as reducing the number of passes, tilling at a shallower depth or avoiding soil inversion. No-tillage is the only type of conservation tillage that appears to promote C sequestration (West and Marland, 2002), although this may only hold true for topsoil (e.g., Luo et al., 2010). Inputs of OC can be maximized by importing organic matter (OM) from other ecosystems, by devoting a larger fraction of the produced biomass to soil application, or by increasing the primary production in the agroecosystem. In all of the cases, the increase in C inputs to the soil typically results in a concomitant increase of C storage in the soil, besides the improvement of other useful agronomic and environmental indicators (Diacono and Montemurro, 2010), such as the increase in microbial biomass and functions (Kallenbach and Grandy, 2011).

Organic agriculture relies on organic matter recycling for the maintenance of soil fertility and crop production, avoiding the usage of synthetic fertilizers and pesticides. As a general trend, organic systems contain more SOC than conventional ones (Mondelaers et al., 2009; Gattinger et al., 2012), despite the large variety of organic farming practices and the width of the underlying principles. On the other hand, SOC benefits from organic farming may be only achieved if the adoption of this management is accompanied by the application of higher C inputs than in conventional treatments (Leifeld and Fuhrer, 2010).

Comprehensive data on sequestration potential by most RMPs have been compiled at a global level and under a number of climates, but no integrated data is available in the specific context of Mediterranean cropping systems. This climate type is found in five different parts of the world (Fig. 1), and it is characterized by seasonal dryness due to hot, dry summers and mild, wet winters, with many of its subtypes being classified as semi-arid. Soil carbon content is typically lower than in temperate areas (e.g. Jones et al., 2005; Chiti et al., 2012). The temporal gap between maximum irradiance and temperature (early summer) and maximum water availability (winter) is responsible for a typically low productivity in Mediterranean rainfed systems. Irrigation, accompanied by intensification of input use, is widely used for this reason. Irrigation affects C dynamics through the simultaneous increase of net primary productivity (and subsequently OM inputs to the soil) and soil respiration, resulting either in a net increase (Wu et al., 2008; Romanya and Rovira, 2011) or decrease (Nunes et al., 2007; Martiniello, 2011) in SOC. Agroecosystem intensification, however, relies on fossil-based inputs, which are associated to high GHG emission.

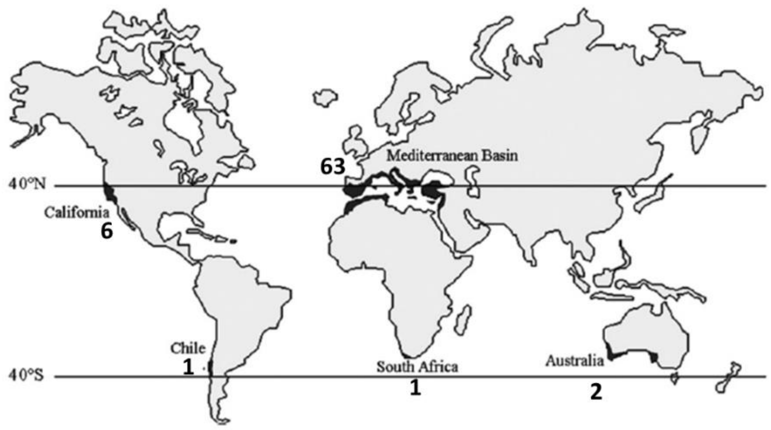

Fig. 1. Areas with Mediterranean climate in the world and number of references selected for each area.

The Mediterranean biome is very biodiverse but highly vulnerable to development pressure and desertification (Underwood et al., 2009). Inadequate agricultural practices are also affecting soil and water quality in regions like the Mediterranean basin (Zalidis et al., 2002). The majority of models coincide in predicting a higher-than-average increase in temperature and decrease in rainfall in most Mediterranean areas (IPCC, 2007), thus reducing water availability (Gibelin and Deque, 2003) and increasing the risk of desertification (Gao and Giorgi, 2008). This scenario is expected to negatively affect crop yield and to increase the risk of yield loss to a much greater extent than in temperate areas (Ferrara et al., 2010; Bindi and Olesen, 2011). Most Mediterranean areas in the world have experienced a decrease in NPP in the 2000-2009 period (Potter et al., 2012). Productivity decrease promotes lower C inputs to soils, while SOC decomposition rate tends to increase with higher temperature (Davidson and Janssens, 2006). As a result, SOC levels could decrease in many Mediterranean areas in the coming decades (e.g. Al-Adamat et al., 2007). These drivers threaten to lower SOC levels below the critical threshold needed for soil fertility, thus highlighting the need for the adoption of adequate management practices, which may induce changes in SOC of greater magnitude than those imposed by climate (Lugato and Berti, 2008; Álvaro-Fuentes and Paustian, 2011; Francaviglia et al., 2012). On the other hand, Mediterranean ecosystems are also interesting experimental models, as global climate change scenarios predict changes in some temperate ecosystems (e.g., higher temperature, summer droughts) which may lead to similarities with Mediterranean climatic conditions (IPCC, 2007; Trnka et al., 2010).

Therefore, the study of management options to maximize SOC levels in Mediterranean agroecosystems is of paramount importance. The present work constitutes the first quantitative review on this subject, comparing a wide range of RMPs (solid organic amendments, land treatment, cover crops, slurry applications, conservation tillage, combined practices and organic management) with conventional management. We conducted a meta-analysis to: i) Estimate the mean change in SOC content and SOC sequestration rate associated to the adoption of RMPs and to the application of different C input rates; ii) Analyze the effect of different organic farming systems and practices on C sequestration, and to: iii) Identify the main sources of uncertainty in the available information.

2. Methods

2.1 Data selection criteria

We collected the available peer-reviewed literature reporting comparisons between RMP and conventional management under Mediterranean climatic conditions. We chose only studies performed in field conditions, in the areas drawn in Fig. 1, excluding laboratory and experimental greenhouse studies. The selected studies included pair-wise comparisons of the performance of RMPs and conventional management, performed under similar pedo-climatic conditions, and after at least 3 or more years of management. Our analysis was also restricted to croplands, including arable crops, orchards and horticulture, but excluding permanent grassland and forests. Studies were selected after searching simultaneously the keywords "Mediterranean", “conventional” and "Soil" in the ISI Web of Knowledge electronic database. The literature cited in the retrieved articles was examined for collecting more studies that could meet the inclusion criteria. When more than one study reported data from the same experiment, only the data from the longest study was included in the analysis.

2.2 Definition of categories

The analyzed data was grouped by category (Table 1). Conventional management (CONV) were used as control groups in most comparisons. CONV treatments typically include synthetic fertilization, conventional tillage and non-use of organic inputs, except crop residues in some cases.

RMPs were grouped in 9 levels, according to their focus on organic inputs or tillage management (Table 1): i) ORG: Organic management. For the purpose of this work, we defined organic farming as a practice that excludes the use of synthetic fertilizers and other synthetic chemicals, e.g., pesticides. These requirements are in agreement with general regulations and recommendations applied in Mediterranean countries and in most of the world. Treatments included in this group may apply some of the practices of the other groups listed below. Those treatments were considered to belong to both the ORG and the other categories, so they were included in all of them; ii) OA: Organic amendments. All kinds of solid, external organic matter sources are included, except municipal solid waste (MSW), sewage sludge (SS) and those amendments applied at rates exceeding 10 Mg C ha^{-1} yr^{-1}, which is considered the maximum agronomic rate usually applied in real agroecosystems; iii) LT: Land treatment of organic residues. The main aim of these experiments was the disposal of organic residues rather than the improvement of soil quality or agronomic performance. This category includes treatments applying MSW or SS at any rate, or other organic inputs applied at rates exceeding 10 Mg C ha^{-1} yr^{-1}; iv) CC: cover crops substituting bare soils, either in herbaceous rotations (substituting bare fallows) or in woody cropping systems (seeded or spontaneous). Cover cropping usually imply some kind of tillage reduction, including cases of no tillage with weeds controlled by mowing; v) Slurry: liquid animal manure, including applications of raw or digested pig or cattle slurry; vi) NT: No-tillage. Soils are subjected to negligible disturbance (usually only narrow slots opened for seed insertion with NT planters), and weeds are controlled with herbicides; vii) RT: reduced tillage. Some kind of tillage exists, but its intensity is lower than in the corresponding CONV treatment. Treatments entering this category include “minimum tillage”, “reduced tillage” and “subsoil tillage”; viii) CMPs: combined management practices. Treatments which combine external inputs (OA) with CC, crop residues, RT or NT. In addition to these RMPs, treatments without any kind of fertilization (named “Unfertilized”) were also compared to conventional, fertilized management.

Table 1. Main features of the categories considered in the analysis.

Category	Full name	Organic input[1]	Tillage type	Observations
CONV	Conventional management	Usually none or CR	Usually conventional tillage	Used as control group
ORG	Organic management	Anyone except MSW or SS	Anyone except NT with herbicides. Usually same as CONV	No synthetic compounds applied
OA	Organic amendments	Compost, manure, AIW	Same as CONV	External inputs applied at <10 Mg C ha^{-1} yr^{-1}
LT	Land treatment	Compost, manure, AIW, MSW, SS	Same as CONV	OA at >10 Mg C ha^{-1} yr^{-1} or MSW/SS at any rate
CC	Cover crops	Cover crops	Conventional, RT, NTM[2]	Cover crops substituting bare fallows
Slurry	Slurry	Slurry	Same as CONV	Raw or digested liquid manures
NT	No tillage	None or CR	NT	Organic C input may differ from CONV
RT	Reduced tillage	None or CR	Reduced, minimum, subsoil tillage	Organic C input may differ from CONV

CMP	Combined management practices	OA and CC/CR	Conventional, RT, NT, NT	OA combined with CC, CR, RT or NT
Unfertilized	Unfertilized	Same as CONV	Same as CONV	No synthetic fertilizer applied

[1] CR: crop residues; MSW: municipal solid waste; SS: sewage sludge; AIW: agro-industrial wastes;

[2] NTM: No tillage by mowing.

ORG-CONV comparisons were more deeply examined in order to determine the impact of different factors on the calculated effect sizes. We studied the effect of 4 variables (Table 2): management intensity, organic input type, crop type and experiment type. Management intensity was grouped in 2 levels: i) Rainfed systems, with no water application; ii) Irrigated systems, treatments receiving some kind of water application. This category also includes 3 comparisons of greenhouse horticulture.

Organic input type refers to the type of organic input which is applied in the organic treatment but not in the conventional one. We categorized the type of organic input in 6 levels: i) Compost (Co), including composts made of vegetal and/or animal sources; ii) Manure (M), including raw solid animal manures; iii) Co+CC, compost + cover crops; iv) M+CC: manure + cover crops; v) CC, cover crops; vi) None, when organic inputs are similar in CONV and ORG treatments. All the treatments actually included in None category lacked any sort of organic inputs, with the only exception of crop residues.

Crop type was classified in 3 levels: i) Cereals, including rotations of differing complexity where cereals were the main crop; ii) Horticulture, including rotations where vegetables were the main crops; iii) Woody crops, including vineyards, olives, citrus and other fruit orchards.

Experiment type was grouped in 2 levels: i) Farm scale, including those studies surveying real organic and conventional farms; ii) Plot scale, including trials conducted under monitored conditions, specifically designed for research purposes.

Table 2. Variables and categories studied in organic management meta-analysis

Variable	Category
Management intensity	Rainfed systems Irrigated systems
Organic input	C (Compost) M (Manure) C+CC (Compost + Cover crops) M+CC (Manure + Cover Crops) CC (Cover crops) None
Crop type	Cereals Horticulture Woody crops
Experiment type	Farm scale Plot scale

2.3 Data management

Data on SOC concentration (g C kg^{-1} soil), SOC stock (Mg C ha^{-1}) and C sequestration rate (Mg C ha^{-1} yr^{-1}) were collected from the treatments included in the database. When data was graphically presented, figures were digitized using GetData software. SOC was calculated from soil organic matter (SOM) data using Mann (1986) relationship (SOC = 0,58*SOM). SOC concentration and SOC stock were estimated for the whole sampled soil profile by calculating the weighted average and the sum of the soil layers, respectively. Only comparisons between full sampled soil profiles were included in the analysis.

C sequestration rate was calculated following equation 1

$$\text{C sequestration rate} = (C_t - C_{t'})/t \qquad (1)$$

Where C_t and $C_{t'}$ represent SOC stocks (Mg C ha^{-1}) at the end and at the beginning of the experiment, respectively, and t refers to the duration of the experiment (years). When data on initial SOC stock were not available, they were equalled to SOC stocks in the conventional treatment, assuming that similar initial C levels existed in RMP and CONV plots. This is justified because only comparisons performed under similar pedo-climatic conditions were selected for the analysis, and we focused only in the relative performance of RMP versus CONV.

Missing data on SOC stocks were calculated following equation 2, for whole sampled soil profiles with k number of layers,

$$\mathrm{C} = \sum_{i=1}^{k} \frac{d_i \rho_i \ \mathrm{SOC}_i}{10} \qquad (2)$$

where d_i and ρ_i are soil depth (meters) and bulk density (BD, Mg m^{-3}) in layer i. Many studies did not report measurements of BD. For the estimation of those lacking data, we re-parameterized Howard et al. (1995) function with data from Mediterranean soils retrieved from the reviewed publications (Fig. 2). The new function (equation 3) is based on 278 pairs of data relating BD with SOC.

$$\rho = 1.84 - 0.443 \log10(\mathrm{SOC}) \qquad (3)$$

We also studied the influence of the amount of C input (Mg C ha^{-1} yr^{-1}) on C sequestration rate. C input rate was calculated from the data provided in the studies when there was enough available information in both the RMP and the CONV groups (143 data sets).

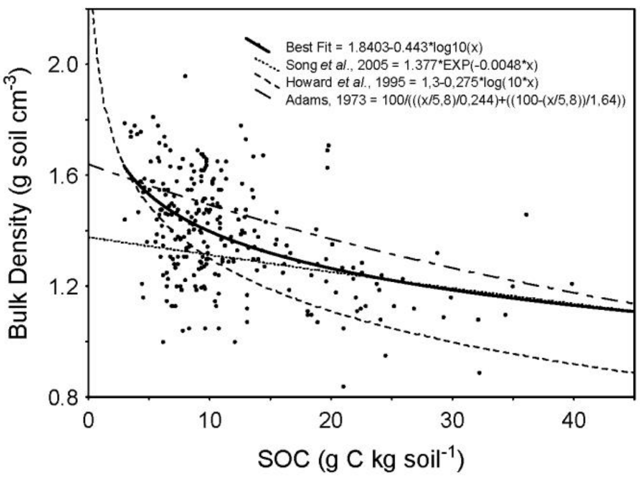

Fig. 2. Bulk density and SOC relationship in Mediterranean cropped soils. The best fit to the data is compared to other functions from literature (Song et al., 2005, Howard et al., 1995, Adams, 1973).

2.4 Statistical analyses

The influence of RMPs on C sequestration was studied through meta-analysis. Paired data comparing SOC concentration (% SOC in soil) and C sequestration rate (Mg C ha^{-1} yr^{-1}) under RMP and conventional management were collected or estimated. When more than one RMP or conventional (control) treatment existed within the same independent study, effect sizes for all possible combinations between RMPs and conventional were calculated. Afterwards, one composite effect size was computed for all combinations of each RMP by computing their mean value, in order to avoid redundancy of the data. Studies were considered to be independent when they were performed on different sites or cropping systems. The SOC database contained 174 data sets from 78 references, while the C sequestration rate database contained 146 comparisons from 67 references (Supplementary materials). The difference is due to the lack of information on the length of the study in some papers, which prevented the calculation of C sequestration rate.

The influence of the mentioned factors on the studied effect sizes was analyzed within the ORG-CONV group. In this case, the amount of information was insufficient to determine the influence of the considered factors using composite data sets. Therefore, paired comparisons were directly analyzed without aggregation. The SOC concentration data base comprised 80 data sets from 30 studies located in Spain, Italy, Greece, Turkey, Israel, California and Australia (11, 6, 4, 1, 1, 5 and 2 studies, respectively). Organic farming C sequestration rate meta-analysis was based on 44 pairwise comparisons from 20 references.

We chose the response ratio (RR) as the effect size unit for SOC content comparisons. RR is defined as the ratio between some measured quantity in the experimental (RMP in our case, $\bar{X}^{RMP}$) and control (conventional, $\bar{X}^{CONV}$) groups ($RR = \bar{X}^{RMP} / \bar{X}^{CONV}$). The formula therefore estimates the proportionate change that results from an experimental manipulation (Hedges et al., 1999). We used the natural log of RR ($L_i = \ln(RR) = \ln(\mathrm{X}^{RMP}) - \ln(X^{CONV})$), because this transformation linearizes the metric and results in a much more normal sampling distribution in small samples (Hedges et al., 1999). SOC concentration (g C kg soil^{-1}) was used for SOC content comparisons when it was available, while SOC stock (Mg C ha^{-1}) was used in the remaining cases. SOC concentration was preferentially chosen for comparisons because it is considered to be a more direct measure of SOC, which is not influenced by soil volume and bulk density estimations (see Discussion).

Raw difference in means was chosen as the effect size for sequestration rate comparisons. This choice is justified because in this case we were interested in knowing the difference between C sequestration in organic and conventional fields, expressed as Mg C ha^{-1} yr^{-1}. Moreover, in many cases C sequestration rate in the control group (conventional plots) was equal to 0. Therefore, the response ratio was not an appropriate measure of effect size for this variable.

In meta-analysis, studies are usually weighted by the inverse of their variance, but this information was not provided in many of the selected studies. Sample size, however, was available in all references. Therefore, in order to include as many studies as possible, but still maintain the philosophy of meta-analysis of giving more weight to larger studies, studies were weighted by sample size (Adams et al., 1997)

$$w'_i = \frac{N_i^{RMP} N_i^{CONV}}{N_i^{RMP} + N_i^{CONV}} \qquad (4)$$

where *w'* refers to the specific weight of the data set, and N^{RMP} and N^{CONV} represent sample sizes in the experimental (RMP) and control (conventional) treatments, respectively. This function results in slightly less efficient weight estimates when all the assumptions of the parametric models are met, but the resampling analyses will be valid even when the parametric analysis is not (Adams et al., 1997). Given that data sets differed in the amount of information that they contained, more weight should be given to data sets that are means of multiple observations (Hungate et al., 2009) and less to those that contain redundant information. Therefore, an additional weighting function was introduced

$$w''_i = n_i / 2k_i \qquad (5)$$

where *n* is the number of independent treatments in the study *ij*, and *k* is the number of data sets finally used in the meta-analysis for that study. Note that in RMP meta-analysis, where data sets are means of multiple observations, *k* is always equal to 1. The specific weight of each data set is therefore given by $w_i = w'_i\, w''_i$

Weighted mean effect sizes of each category were calculated, with bias-corrected 95% confidence intervals (CIs) generated by a bootstrapping procedure (10000 iterations) (Adams et al., 1997), using MetaWin software (Rosenberg et al., 2000).

3. Results

The 174 data sets retrieved for the meta-analysis belonged to 79 different publications. Detailed information about the selected studies is provided in the appendix (Table A.1). The category with most information was organic management (ORG), with 53 data sets from 30 publications (Table 3). Overall, initial C was reported in 55% of the data sets, bulk density in 46.5%, C input in 51.3% and coarse (> 2 mm) fraction in only 4.1%. Mean experiment duration was 7.9 years and sampling depth 25.7 cm. Sampling depth was highest in NT category (33.8 cm) and lowest in ORG category (19.4 cm).

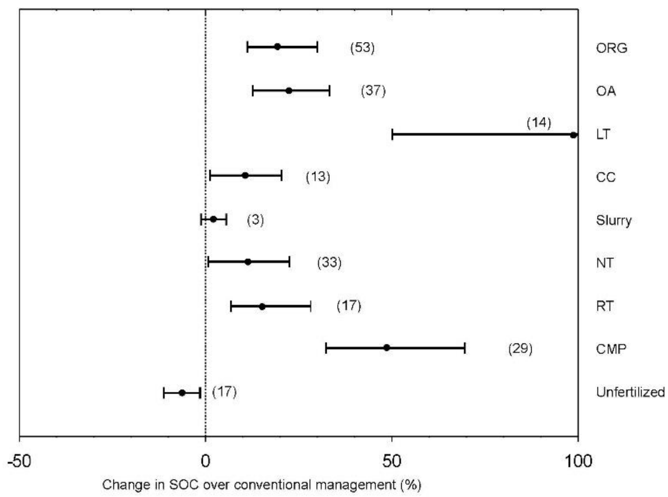

Fig. 3. Effect of different recommended management practices (RMPs) on soil organic carbon (SOC) in units of percent change from the control (conventional management). ORG: organic management; LT: land treatment (urban wastes and C inputs exceeding 10 Mg C ha^{-1} yr^{-1}); OA: organic amendments; CC: cover crops; Slurry: liquid manures; NT: no tillage; RT: reduced tillage; CMP: combined management practices (OA combined with CC, CR, RT or NT); Unfertilized: no organic or synthetic fertilizers are applied. Error bars represent confidence intervals at 95 %. Number of data sets is given in parentheses.

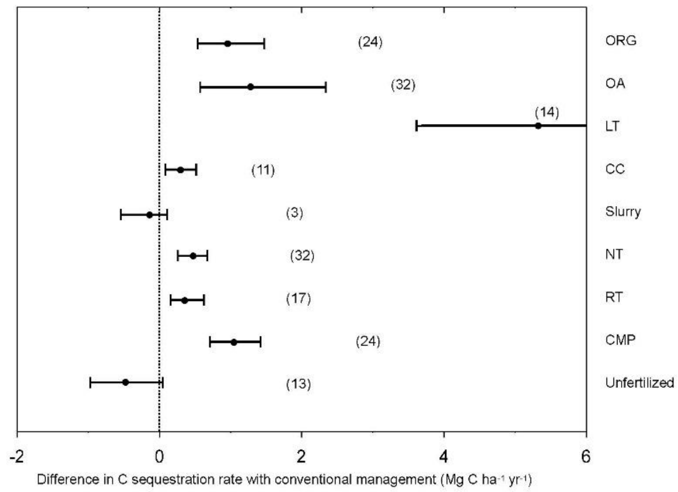

Fig. 4. Effect of different recommended management practices (RMPs) on C sequestration rate, compared to conventional management. ORG: organic management; LT: land treatment (urban wastes and C inputs exceeding 10 Mg C ha^{-1} yr^{-1}); OA: organic amendments; CC: cover crops; Slurry: liquid manures; NT: no tillage; RT: reduced tillage; CMP: combined management practices (OA combined with CC, CR, RT or NT); Unfertilized: no organic or synthetic fertilizers are applied. Error bars represent confidence intervals at 95 %. Number of data sets is given in parentheses.

Most of the RMPs studied were associated to a significant increase in SOC, when compared to conventional management (Fig. 3). RMPs were classified according to their focus on organic matter inputs or on tillage management. The influence of organic matter input is shown by LT, OA, CC and Slurry categories (Figs. 3 and 4). On average, SOC content increased by 98.2% (Fig. 3) and C sequestration rate by 5.29 Mg C ha^{-1} yr^{-1} (Fig. 4) in LT treatments. The application of organic amendments at agronomic rates (OA) increased SOC by 23.5% and C sequestration rate by 1.31 Mg C ha^{-1} yr^{-1}. In CC category, where the organic matter input is always produced within the system, the average increases in SOC were reduced to 10% (Fig. 3) and C sequestration averaged 0.27 Mg C ha^{-1} yr^{-1} (Fig. 4). In Slurry category the differences with conventional management were not significant. C input rate had a significant correlation with C sequestration rate in the reviewed studies (R, Spearman = 0.74, p<0.001, Fig. 5). In many cases, SOC sequestration was higher than 50% of added C input, and in some cases it was even higher than 100%.

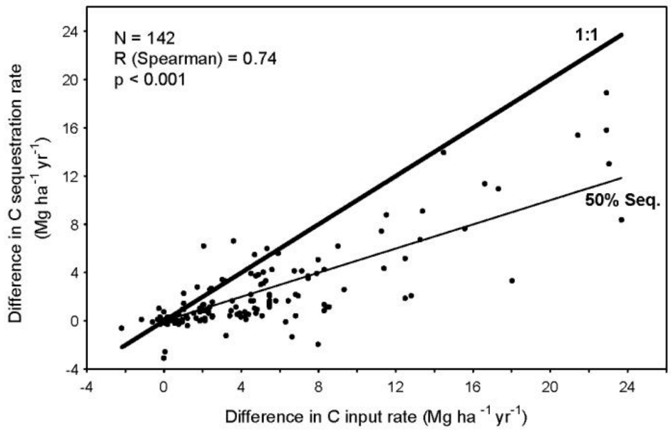

Fig. 5. Difference in C sequestration rate versus difference in C input rate of recommended management practices compared to conventional management

Both of the studied conservation tillage categories enhanced C sequestration. NT showed an average increase of 11.4% in SOC resulting in a C sequestration rate of 0.44 Mg C ha^{-1} yr^{-1}, whereas under RT, SOC content increased by 15% and C sequestration rate by 0.32 Mg C ha^{-1} yr^{-1}. The relatively lower effect of NT on SOC is influenced by the inclusion of 5 woody crops treatments. SOC increased by 18.2% (N = 28) under NT in herbaceous crops, but decreased by 22.9% (N = 5) in woody crops. CMP is a mixed category, where organic matter inputs and conservation tillage practices are simultaneously applied. CMPs promoted an increase of 49.2% in SOC, and enhanced C sequestration rate by 1.11 Mg C ha^{-1} yr^{-1}.

SOC concentration was increased by 19.2% and C sequestration rate by 0.97 Mg C ha^{-1} yr^{-1} in organic treatments. Figures 6 and 7 show the results of the meta-analysis of ORG-CONV comparisons. SOC increment in organic systems was greater under irrigation than under rainfed conditions (25% vs. 13% increase over conventional, respectively). When analyzed by crop type, the data shows that the best performing organic group is horticulture, where SOC is increased by 48% (Fig. 6). Groups in Irrigation and Crop Type categories differed in the intensification of C input rate. Mean C inputs (additional C input over conventional) were 4.8 and 3.2 Mg C ha^{-1} yr^{-1} in irrigated and rainfed systems, respectively, and 6.1, 2.5 and 3.1 Mg C ha^{-1} yr^{-1} in horticulture, cereals and woody crops, respectively.

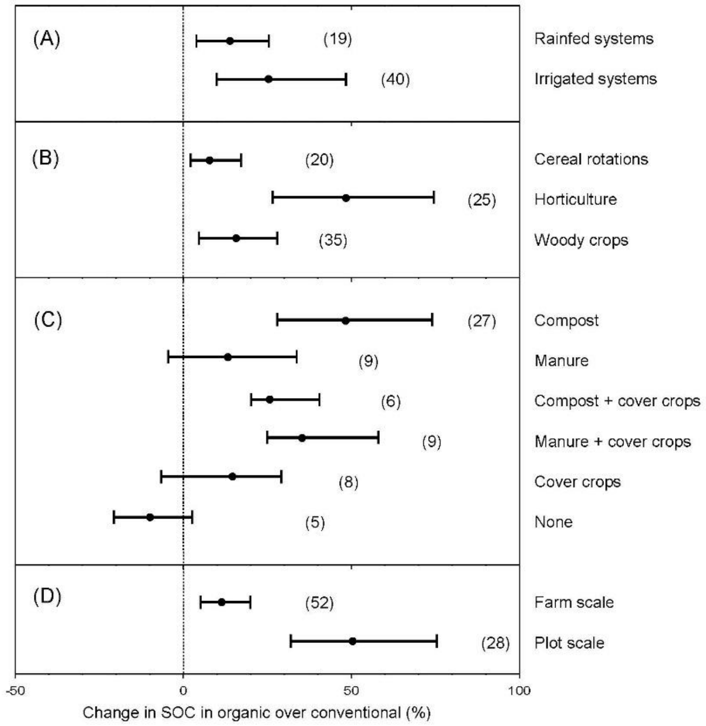

Fig. 6. Percent change in SOC in organic treatments, compared to conventional ones. Systems are grouped by management intensity (A), crop type (B), organic input type in the ORG treatment (C) and experimental approach (D). Error bars represent confidence intervals at 95 %. Number of comparisons is given in parentheses.

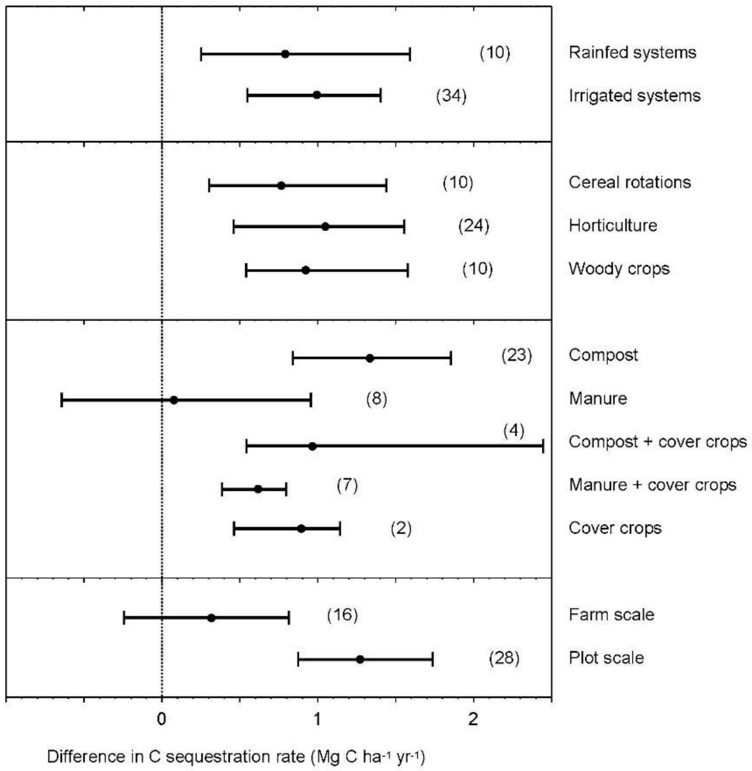

Fig. 7. Difference in C sequestration rate between organic and conventional management. Systems are grouped by management intensity (B), crop type (C), organic input type (D) and experimental approach (E). Error bars represent confidence intervals at 95 %. Number of comparisons is given in parentheses.

The type of organic input employed in the organic system also influences the differences found between systems. Compost, either applied alone or in combination with cover crops, is the input associated to highest increases in SOC (48% and 26.2% for compost alone and mixtures with cover crops, respectively, Fig. 6) and in C sequestration rate (1.32 and 0.97 Mg C ha^{-1} yr^{-1} for compost and mixtures, respectively, Fig. 7). Manure application obtains poorer results, being the increase in C sequestration rate over conventional non-significant when this amendment is applied alone, although the sample size is lower in this case (9 paired comparisons versus 27 in Compost alone). When manure is combined with cover crops, the increase is higher and significant (35.8% and 0.62 Mg C ha^{-1} yr^{-1}). We have only two paired data of cover crops used alone. This category, however, was also studied in the general meta-analysis, where the number of studies was larger (see Figs. 3 and 4). Finally we have found some organic treatments where no organic inputs were applied, or at least no more than in their conventional counterpart. These treatments were the only ones with a lower SOC level than conventional, although the differences were not significant.

The last variable studied was the type of experimental approach. The increase in SOC concentration and sequestration rate in organic plots was much more pronounced in plots of controlled experiments (51.6% and 1.28 Mg ha^{-1} yr^{-1}) than in real farms (11.4% and 0.31 Mg ha^{-1} yr^{-1}, being the latter non-significant).

4. Discussion

Recommended management practices studied influence SOC dynamics mainly by modifying C inputs, tillage or both of these factors.

4.1 Organic matter inputs and C sequestration.

The relationship between the application of OM to the soil and the gains in SOC is very clear in the reviewed data (Fig. 5). However, there was a very high variability, due to the variety of practices and agro-ecological conditions included. Largest increase in SOC was achieved by Land Treatment (LT), where very high amounts of organic inputs were applied (usually higher than 10 Mg C ha^{-1} yr^{-1}). This practice is obviously not scalable due to the use of very high OM inputs, but its strong effect on SOC suggests that cultivated Mediterranean soils are far from their potential for SOC storage, i.e., from soil C saturation (West and Six, 2007).

A very high sequestration of C was found in soils under Organic Amendments (OA) treatments (1.34 Mg C ha^{-1} yr^{-1}). OA category comprises many kinds of external sources of organic matter, including compost, manures and agro-industrial wastes. The external origin of many of these materials allows the intensification of the rates applied in the reviewed experiments (on average 3.45 Mg C ha^{-1} yr^{-1} more than conventional), even if we have only included treatments receiving less than 10 Mg C ha^{-1} yr^{-1}. These amounts of residues could only be self-produced in very productive systems with high residue production, which are not predominant in Mediterranean areas. In the other cases, they rely on imports from other systems.

This level of intensification is not possible in Cover Crop (CC) category, where no external C is imported to the system. This limitation is reflected in the average increases in SOC (10%, Fig. 3) and C sequestration rate (0.27 Mg C ha^{-1} yr^{-1}, Fig. 4), which are lower than in the previous two categories, although the differences with conventional are still significant. These numbers are lower than those obtained for Spain in a recent meta-analysis (González-Sánchez et al., 2012), where cover crops were associated to mean sequestration rates of 1.59 Mg C ha^{-1} yr^{-1}. In any case, the potential of this practice, relies in that it contributes to the soil C balance with additional biomass produced within the cropping system. It also minimizes soil disturbance and efficiently protects the soil against erosion, which is also an important process in SOC depletion (Lal, 2003). Adequate designs allow the cultivation of cover crops in substitution of bare soil, while maintaining yields of commercialized crops (Gabriel and Quemada, 2011; Ruiz-Colmenero et al., 2011). In perennial woody systems such as olive groves, vineyards or almond orchards, the improvement of soil quality and the reduction in erosion associated to cover crops are very high (Castro et al., 2008; Moreno et al., 2009, Gómez et al., 2011).

The way in which wastes of other agricultural and livestock activities are re-used can also affect SOC dynamics. Thus, “Slurry” category shows the lowest performance, being non-significantly different from conventional management (Figs. 3 and 4). The absence of a clear effect of slurries on SOC is in accordance with studies performed in other climates (Rochette et al. 2000). Peñuelas et al. (2009) studied the evolution of soil nutrients and other parameters along four decades in a Mediterranean area intensively fertilized with pig slurries. They found that, although soil C increased non-significantly in the second decade, it returned afterwards to previous levels. Liquid organic materials are usually highly mineralized, with a very low and easily decomposable amount of C, and a very low C:N ratio (Plaza et al., 2004; Meijide et al., 2009). This results in a high N availability for microorganisms, which obtain their added energy requirements through the oxidation of the native organic C. In consequence, slurries usually do not promote SOC accumulation (Rochette et al. 2000; Peñuelas et al., 2009).

In organic management meta-analysis, organic amendments have been separated in raw and composted materials. The results point at a higher C sequestration gain with compost than with raw manure (Figs. 6 and 7). This could be due to the more stabilized forms of C present in compost than in raw manures or other raw organic materials. C conservation efficiency by composting, however, should be compared with the use of raw materials taking also into account C losses during the composting period (Mondini et al., 2007; Sánchez-Monedero et al., 2008; Bernal et al., 2009). A full assessment of composting should also include other appropriate indicators, such as the emission of other GHG (Sánchez-Monedero et al., 2010) and a germination index measuring phytotoxic effects (Sánchez-Monedero et al., 2008).

Soil application of organic wastes is a key tool to close nutrient cycles. The positive effect on SOC observed in our analysis further supports the use of these materials as fertilizers. Soil C accumulation implies an improvement in soil quality and a protection against erosion, which are especially important effects in desertification-prone Mediterranean agroecosystems. From a climate change mitigation approach, C sequestration adds to other benefits such as the reduction in the use of synthetic fertilizers and the GHG emissions related to their production. In addition, direct N_2O emissions from the soil could possibly be lower for organic than for synthetic fertilizers under Mediterranean conditions (Aguilera et al., 2012). A further benefit from the N cycle perspective is the sequestration of reactive N (N_r) in the SOM; sequestered N_r represents ca. 10% of sequestered C (Kong et al., 2005).

As in the specific case of composting, a full global warming potential (GWP) accounting is needed to evaluate different residue management strategies applied to particular agroecosystems. It is important to take into account that the net transfer of C between the atmosphere and the soil depends on the alternative fate of the residue (Powlson et al., 2011). In addition, estimations of the potential sequestration by these practices must consider the amounts of residues produced and the distribution of the sources in the landscape (Sommer et al., 2009). The greatest potential is in the use of materials that are nowadays frequently burned in the field (pruning residues and straw) or constitute an environmental problem (olive mill and other agro-industrial wastes, urban wastes). The agronomic use of urban wastes requires a careful separation or pre-treatment at source to avoid potential toxic pollutants.

4.2 Tillage

Practices focusing on tillage management (No Tillage, NT, and Reduced Tillage, RT) face the same constraint as CC, because their sources of OM are also internal and therefore limited by system productivity. Nevertheless, SOC content and C sequestration rate are significantly higher than in conventional soils for both conservation tillage categories. NT shows an average increase of 11.2% in SOC and of 0.44 Mg C ha^{-1} yr^{-1} in C sequestration rate. The response under RT was lower but very similar, with average increases of 0.32 Mg C ha^{-1} yr^{-1} over conventional management. The results agree with those obtained by González-Sánchez et al. (2012), who found C sequestration rates of 0.72 and 0.29 Mg C ha^{-1} yr^{-1} under NT and -0.01 and 0.43 Mg C ha^{-1} yr^{-1} under minimum tillage in continental and maritime Mediterranean areas of Spain, respectively. These data can be compared to global figures compiled by Smith et al. (2008), where mitigation potential by tillage and residue management practices was established in 0.15, 0.51, 0.33 and 0.7 Mg C ha^{-1} yr^{-1} in cool-dry, cool-moist, warm-dry and warm-moist climatic zones.

Stratification (Franzluebbers, 2002) is higher when tillage is reduced due to less soil disturbance, a process also widely reported in Mediterranean soils (e.g., Hernanz et al., 2009). Stratification is positive for erosion control and water and nutrient conservation, but makes sampling depth to be particularly relevant when determining net changes in SOC, because surface gains may be accompanied by losses in deep layers (see Gaps section). Another important confounding factor in the comparison of tillage methods is differing C input amounts between both treatments of a paired comparison. This is illustrated by the large difference between the performance of herbaceous and woody NT systems, with a highly significant enhancement of SOC (18.2%, N = 28) in the first case and a decrease in SOC in the latter (-22.9%, N = 5). In Mediterranean herbaceous systems, no-tillage practices are usually associated to higher C inputs to the soil, mainly because straw is retained instead of being removed or burned (Moreno et al., 2006; Muñoz et al., 2007; Melero et al., 2009; Sombrero and de Benito, 2010; Mazzoncini et al., 2011). Only in two studies, positive C sequestration rates were observed in NT treatments receiving slightly lower inputs than their conventional pairs (Alvaro-Fuentes et al., 2009; López-Bellido et al., 2010). The opposite trend occurs in woody crops, where NT using herbicides is associated to lower weed biomass than under conventional tillage, resulting in lower C inputs to the soil (e.g., Castro et al., 2008; Steenwerth and Belina, 2010). Therefore, the relative amount of C input is highly relevant for C sequestration also under NT practices.

Tillage management can also influence other areas of the GHG balance of cropping systems. Tillage reduction or suppression usually implies a decrease in CO_2 emissions associated to farm operations, mainly driven by a lower fuel consumption (Govaerts et al., 2009), although there may be an increase in emissions from other inputs, such as lime (Robertson et al., 2000) and herbicides. In addition, no-tillage can be associated to higher N_2O emission in Mediterranean cropping systems (Aguilera et al., 2013). This effect has also been observed under temperate climate, although it may disappear in the long term (Six et al., 2004).

4.3 Combined practices

Combined Management Practices (CMP) effectively enhanced SOC in Mediterranean cropping systems, with ca. 50% increase in SOC levels over conventional management. These high increments suggest that the combination of external organic inputs (OA) with RMPs based on internal recycling and conservation of organic matter (crop residues, CC, RT and NT) is an adequate way of optimizing agroecosystem resources for SOC and soil quality maximization. Woody crops have a very high potential for this optimization, as cover crops can be combined with the application of pruning residues and agro-industrial wastes, such as olive mill wastes. In kiwifruit and apricot orchards under Mediterranean climate, mean annual C inputs were increased from 1.5 to 9 Mg ha^{-1} in the local and soil protecting (SP) management, respectively (Montanaro et al., 2009). This was achieved by combining compost application (5 Mg C ha^{-1}) with the implementation of cover crops, no-tillage, and mulching of pruning residues. Although SOC did not change after 4 years, yields increased by 28-50% in SP treatments despite receiving much less mineral fertilization.

4.4 Unfertilized treatments

SOC levels in the Unfertilized group of the general meta-analysis (Fig. 3) were slightly, but significantly lower than in conventional treatments, with a reduction of 6.8%. This effect was also observed (although differences were not significant in this case) in organically managed soils receiving no organic amendments, or similar amounts to their conventional, fertilized control groups (Fig. 6). Higher SOC levels under conventional fertilization than in unfertilized plots are probably due to higher plant growth, which results in increased C inputs to the soil. This result agrees with the global data compiled by Lu et al. (2011), where SOC increased by 3.5% in N fertilized agricultural systems. These authors noticed that N addition substantially increased C inputs to soil systems, but this resulted in only minor changes in soil C storage. N addition altered plant C allocation, with more to above-ground growth, but N-induced change in C storage was only correlated with below-ground plant growth.

4.5 Organic management systems

SOC was increased by 19.2% and C sequestration rate by 0.97 Mg C ha^{-1} yr^{-1} in organic treatments (Figs 6 and 7, respectively). In SOC meta-analysis, weighting was very unevenly distributed due to large differences in sample sizes. In the study by Campos-Herrera et al. (2010), sample size was 40 for each treatment. The three data sets retrieved from this reference summed up 41% of all the weight in this meta-analysis. When these data were excluded from the analysis, average increment in SOC was 26.5% (k = 50) under organic management.

SOC increment in organic over conventional was affected by irrigation and crop type, with largest increases in most intensively managed cropping systems. This is due to the fact that intensification in organic systems means increases in the amount of organic matter applied to soils. Thus, largest C input rates were applied in the organic groups with the largest observed effect sizes. For example, average C input rates of 6.1, 2.5 and 3.1 Mg C ha^{-1} yr^{-1} promoted increases in SOC of 48%, 8% and 15.5% in horticulture, cereals and woody crops respectively.

The analysis of C sequestration at the plot and farm scales shows a large gap between the optimal and the actual C sequestration in Mediterranean organic farming systems, in agreement with the findings from the global meta-analysis performed by Leifeld and Fuhrer (2010), where SOC increased by 3.24% (±1.76, N = 27) per year in plot experiments and by 1.07% (±0.28, N = 27) in farm-scale comparisons, although the differences were only significant in the second case ($P < 0.05$). In our study, the change over conventional was also greater in plot experiments (51.6%) than in real farms (11.4%), and it was significant in both cases. The differences between both groups seem to be mainly driven, as with the other categories, by lower application of OM in real farms, although many studies did not report the amounts of C inputs applied. On average, 4.3 Mg C ha^{-1} yr^{-1} (n = 23) were applied in plot experiments and 1 Mg C ha^{-1} yr^{-1} in real farms (n = 3). In plot experiments, OA (compost or manure) were applied in 93% and CC in 35% of the organic treatments. In farm comparisons, OA were present in 46% and CC in 23% of the organic treatments (11% of the total were combinations OA+CC), while in 10% neither type of input was present, and 25% of the data sets did not include enough information to be classified. These differences in the degree of adoption of RMPs between experimental plots and real farms suggest that there are constraints limiting the use of RMPs in real cropping systems. These constraints can be physical. When intensification of C input relies on external sources, they may not be locally available. When it is based on internal resources, competence with other uses may arise (feed, fuel, wood). Constraints can also be economical, for example a high price of commercial composts, or a high cost of farming operations such as composting, land spreading of organic matter, or cover crop cultivation. Last, constraints can be social, as farmer may be unaware of the management practices that could result in SOC increases, and about the benefits derived from them. More research should be done to address these constraints and develop adequate policies. Understanding the potential for C sequestration under organic farming would probably require a top-down approach based on the availability of organic matter for its application to soils.

4.6 Gaps in the available information

We identified a number of sources of uncertainty in the analyzed information. In this section, we describe those information gaps, identify possible sources of bias and suggest some ways to improve the accuracy of the experimental methodologies.

4.6.1 *Bulk density (BD) is not provided in many cases*

The calculation of SOC stock normally requires information on SOC concentration, BD and sampling depth. Many of the reviewed studies lacked BD information; therefore a pedotransfer function based on Mediterranean data was developed to estimate this parameter from SOC concentration data. Even so, the uncertainty is very high in this kind of estimations due to the influence of other factors apart from SOC concentration on BD, such as C input type, tillage or soil depth. For example, Tejada and González (2006) found that BD decreased with the use of composts, whereas it increased with the application of sugar beet vinasse, despite both types of inputs had a positive effect on SOC. These differences underline the need for direct BD measurements or more precise pedotransfer models.

4.6.2. *SOC stock calculation is biased by changes in bulk density*

The estimation of SOC stock may be biased even when BD is directly measured in the soil, because management-induced BD changes modify the mass of soil sampled when measurements are taken at a fixed soil depth (Ellert and Bettany, 1995; Lee et al., 2009). As a general trend, changes in BD are negative in the case of organic input applications (e.g. Celik et al., 2009; Efthimiadou et al., 2010) and positive in the case of tillage reductions (e.g. Gómez et al., 1999; Hernanz et al., 2009, although not always: Alvaro-Fuentes et al., 2008), leading to an underestimation of SOC stock in the first case and to an overestimation in the second. In Celik et al. study (2004), the treatment receiving the largest dose of compost (25 Mg) was also the one where SOC increased most, but BD decreased most. A similar process occurred with the long-term organic treatment in Efthimiadou et al. (2010) study. In both cases, the reduction in BD caused SOC stock estimated with the data provided by the authors to be lower for the alternative treatment, despite having the highest SOC content. By contrast, similar SOC contents were measured in conventionally tilled and NT olive orchards (Gómez et al., 1999), but estimated SOC stocks were 34% higher for the NT treatment due to an increase in BD. These errors can only be overcome if SOC stocks are calculated in terms of equivalent soil mass (ESM) (Ellert and Bettany, 1995). This approach was only followed in one of the analyzed studies (Hernanz et al., 2009), where SOC stocks were calculated employing the maximum ESM method. However, Lee et al. (2009) demonstrated that only the original ESM method, where initial soil mass is used as ESM, results in accurate estimations of SOC stock changes. We strongly support using the original ESM method instead of the fixed soil depth for SOC stock estimation.

4.6.3. *Coarse soil fraction (> 2 mm) is usually neglected*

Non-accounting for coarse fraction is an important source of error in SOC stock estimations, especially in arid soils, where this fraction is usually significant (Throop et al., 2012). It implies an overestimation or underestimation of SOC stock depending whether the coarse fraction is mainly composed by stones or by organic debris, respectively. The quantities of C stored in stony fractions remain usually unchanged, unlike in the < 2 mm fraction. A high fraction of stones in the soil is very common in many Mediterranean croplands, especially in woody crops, which are often cultivated in mountainous or marginal land. For example, Ramos et al. (2007) measured over 65% of gravels in vineyards, which drastically influenced SOC content estimation. Other examples are 30% (Alvarez et al., 2007) and 26-55% (Sierra et al., 2001) in olive orchards of South and North Spain, respectively, 45% in cereal rotations of South-Africa (Agenbag and Maree, 1989), and up to 90% of coarse elements in almond orchards of Southern Spain (Quine et al., 1999). When the rejected > 2 mm fraction is composed by organic debris, the total organic C content in the system is underestimated, although this is usually accepted due to less stable form of debris C.

4.6.4. Experiment length is usually short or medium-term (average 8.7 years)

Changes in C stocks due to changes in management practices should be observed in large temporal scales. A high short-term variability may be induced by climatic factors such as interannual or seasonal precipitation patterns (Chou et al., 2008). Long-term studies are therefore more reliable, but they have been scarce until very recently in Mediterranean areas. 70% of the reviewed experiments lasting for 10 or more years were published from 2007 to 2011 (Fig. A.1).

4.6.5. *Sampling depth is commonly insufficient (average 23 cm).* A number of the analyzed studies shows that changes in SOC stocks occur deeper than 20 cm (Carbonell-Bojollo, 2010), and even deeper than 60 cm (López-Bellido et al., 2010), but a very small portion of the studies includes sampling in deep layers (Fig. A.2). In principle, the incorporation of organic matter to the soil can increase SOC to a depth of 40 cm or more (Luo et al., 2010), especially in the long term, but also in the short term. In consequence, studies on organic inputs tend to underestimate C sequestration when sampling depth is small (e.g. 20 cm). Shallow sampling was often the case in organic farming studies, where average sampling depth was just 19.2 cm. (Table 3). By contrast, recent data compilations suggest that no-tillage practices may only be inducing a redistribution of C in the soil profile, with a net accumulation in the surface layers and a net loss in deeper layers (Manley et al., 2005; Baker et al., 2007; Angers and Eriksen-Hamel, 2008; Luo et al., 2010).Therefore, shallow sampling under RT or NT may imply an overestimation of SOC gain in those treatments, given that losses occurring in deep layers are not accounted for. In our study, mean sampling depth was 33.8 cm in NT studies and 27.2 cm in RT studies (Table 3). Although these numbers are among the highest of the studied categories, it is still insufficient to make sound conclusions from the data provided. There was only one reference (López-Bellido et al., 2010) with sampling deeper than 40 cm. In this study, C sequestration at depth under NT was observed under some rotations (continuous wheat and wheat-faba bean), even in the 60-90 cm soil layer. The study, however, took place on a vertisol, where cracks can feed SOC to deep layers. The differences induced in other treatments in comparison with those from conventional tillage were not significant neither in the whole profile (0-90 cm), nor in the surface (0-15 cm).

4.6.6. *Full C input estimations are very scarce.* According with our analysis of the available data, C input is the main driver of the changes in SOC produced after the adoption of RMPs. However, information on C input was only provided in 42.2% of the data sets studied (Table 3), and in most cases this information was incomplete. Thus, usually only the amount of C applied in the external C input was provided, while the internal sources of C were ignored.

5. Conclusions

Increasing soil organic carbon (SOC) is a key process in both mitigation and adaptation strategies to climate change. It has a particular relevance in Mediterranean agroecosystems, where soils usually have a low SOC content and are very vulnerable to desertification. Our analysis of the available data indicates that SOC is highly sensitive to changes in management under Mediterranean conditions, as most recommended management practices significantly increased SOC when compared to conventional management. Very high SOC increases are achieved by the application of external organic inputs to the soil (land treatment, organic amendments), suggesting a high potential for C storage in the soil. SOC is also increased by internal organic inputs and the reduction in soil disturbance (cover crops, reduced tillage, no tillage). The combination of organic inputs and reduced disturbance (combined management practices) is the most promising strategy to maximize SOC levels. Slurry applications only maintained SOC at the same level as conventional, while in unfertilized treatments a slight decrease was observed.

C sequestration is effectively promoted by organic farming practices in Mediterranean cropped soils. This relative increase of SOC sequestration over conventional practices is more marked in more intensive cropping systems, where differences in C inputs are higher. The relative sequestration is also much higher in experimental plots than at the farm scale. These findings suggest that best organic practices are not widespread enough in real organic farms, hindering the development of the full C sequestration potential of organic management.

Important information gaps were detected in the reviewed studies. In particular, SOC stocks, and subsequently C sequestration rate estimations may be biased because soil volume per unit surface is altered by changes in bulk density and coarse fraction. Small sampling depth and short experiment duration further increase the uncertainty of the data.

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Appendix A

Table A.1 Characteristics of the studies used in the meta-analysis

Reference	Location	Crop system	C input[1]	Tillage[2]	Categories[3]	Management intensity	Duration[4] (years)	Max. depth (cm)	SOC Pairs[5]	Sample size	Observations
Agenbag and Maree, 1989	Langewens, South Africa	Wheat, wheat-pasture	CR burnt, CR	T, RT, NT	RT, NT	Rainfed	10	10	4	4	Coarse fraction quantified
Albiach et al., 2001a	Moncada, Valencia, Spain	Lettuce-chard	None, SS	T	LT, Unfert.	Irrigated	10	30	2 (7)	4	
Albiach et al., 2001b	Moncada, Valencia, Spain	Lettuce-chard	None, MSW-C, C, M, SS	T	OA, LT	Irrigated	5	15	2 (4)	4	
Altieri and Esposito, 2008	Doglio, Perugia, Italy	Olive	CC, OMW-C	NTM	OA	Irrigated	5	20	1 (2)	3	CC is used in CONV
Álvaro-Fuentes et al., 2008	Selvanera, Lleida, Spain	Wheat-barley-rapeseed	CR	T, RT, NT	RT, NT	Rainfed	18	40	2 (3)	3	
	Agramunt, Lleida, Spain	Wheat-barley	CR	T, RT, NT	RT, NT	Rainfed	15	40	2 (3)	4	
Álvaro-Fuentes et al., 2009	Aula Dei, Zaragoza, Spain	Barley, barley-fallow	CR	T, RT, NT	RT, NT	Rainfed	16	30	4	3	Aerial biomass C input (average of the last 5 years)
Andrews et al., 2002	San Joaquin Valley, California, USA	Vegetables (many), cotton	None, M-C, AIW-C, CC	T	OA, CC, CMP	Irrigated	3	15	6	6	Initial BD not availble. It has been assumed to be unchanged
	San Joaquin Valley, California, USA	Vegetables (many), cotton	None, M, M-C	T	ORG, OA	Irrigated	3, 10	15	1 (3)	6	
Antoniadis et al., 2010	Volos, Greece	Cotton	None, SS		LT	Irrigated	4	25	1 (3)	4	BD and control data from Samaras, 2008
Aranda et al., 2011	Sierra Mágina, Jaén, Spain	Olive	None, M, CC	T, NTM	ORG, CMP	Rainfed		46	2	3	Coarse fraction quantified
Badalucco et al., 2010	Ripagnola, Puglia, Italy	Vegetables (many)	None, C	T, RT	CMP	Irrigated	7	15	1	5	0-5 cm soil layer not quantified
Benitez et al., 2006	Pedroches, Córdoba, Spain	Olive		T, NT,	ORG, CC			20	1 (8)	3	0-10 cm soil layer not quantified

				NTM							
	Montes Orientales, Granada, Spain	Olive		T, NT, NTM	ORG, CC, NT			20	2 (5)	3	0-10 cm soil layer not quantified
Bessam and Mrabet, 2003	Sidi El Aydi, Settat, Morocco	Wheat rotations	CR	T, NT	NT	Rainfed	4	20	1	3	
Campos-Herrera et al., 2010	Oja-Tirón Valley, La Rioja, Spain	Arable, orchard, vineyard			ORG			18	3		
Canali, 2003	Eastern Sicily, Italy	Citrus			ORG	Irrigated		25	1	27	
Canali et al., 2004	Lentini, Sicily, Italy	Citrus	None, M-C, AIW-C, M		ORG, OA	Irrigated	6	25	1 (3)	3	
Canali et al., 2009	Catania, Sicily, Italy	Citrus			ORG, CMP	Irrigated		25	1	13	
Carbonell-Bojollo et al., 2009	Montoro, Córdoba, Spain	Olive	None, OMW-C		OA	Rainfed	3	60	2	6	
Castro et al., 2008	Finca Salido Bajo, Jaén, Spain	Olive	None, CC	T, NT, NTM, RT	NT, CC	Rainfed	28	30	2 (3)	7	
Celik et al., 2004	Adana, Turkey	Wheat-pepper-maize	None		OA, Unfert.	Irrigated	4	30	2 (4)	3	
Clark et al., 1998	SASF, Sacramento Valley, California, USA	Tomato-safflower-beans-maize	CR, CC, M	T	CC, ORG, CMP	Irrigated	8	30	2 (4)	8, 16	
Cookson et al., 2006	South-western Australia	Wheat rotations	None, C	T, RT	RT, ORG, OA	Rainfed	>8	5	1 (2)	12, 4	RT used in CONV
Deria et al., 2003	South-western Australia	Wheat rotations	None, C	T, NT	NT, ORG, OA, Unfert.	Rainfed	>3	10	7	4	NT used in CONV
Drinkwater et al., 1995	Central Valley, California, USA	Tomato	None, CR, CC, M	T	ORG, CMP	Irrigated	7	20	1	12	
Efthimiadou et al., 2010	Prespa, NW Greece	Beans	None, M	T	ORG, OA	Irrigated	2	12	1 (2)	10	
Fernández et al., 2009	La Higueruela, Toledo, Spain	Barley	None, SS	T	LT	Rainfed	3	15	1 (8)	4	Sampling depth not specified. It has been equaled to tillage depth
Fernández-Ugalde et	Santacara, Navarra, Spain	Barley	CR	T, NT	NT	Rainfed	7	30	1	4	

al., 2009											
Gacía-Gil et al., 2000	La Higueruela, Toledo, Spain	Barley	None, M, MSW-C	T	OA, LT, Unfert.	Rainfed	9	20	1 (4)	4	
García-Ruiz et al., 2009	Pegalajar, Jaén, Spain	Olive	None, M, CC	NT, NTM	ORG, CMP	Irrigated, rainfed	8	10	1	3	SOC data from pers. Comm
	Puente Tablas, Jaén, Spain	Olive	None, M	T, RT	ORG, OA	Irrigated	7	10	1	3	SOC data from pers. Comm.
	Puente de la Sierra, Jaén, Spain	Olive	None, M, CC	NT, NTM	ORG, CMP	Irrigated	8	10	1	3	SOC data from pers. Comm.
Gómez et al., 1999	Santaella, Córdoba, Spain	Olive	None	T, NT	NT	Rainfed	13	9	1	10	
Gómez et al., 2009	La Conchuela, Córdoba, Spain	Olive	None, CC	T, RT, NT	NT, CMP	Irrigated	7	10	2	6	
Herencia et al., 2007	Torres-Tomejil, Alcalá del Río, Sevilla, Spain	Vegetables (many)	None, C, CR		ORG, CMP	Greenhouse	9	15	3	8	
Herencia et al., 2011	Torres-Tomejil, Alcalá del Río, Sevilla, Spain	Vegetables (many)	None, C, CR		ORG, CMP	Irrigated	10	15	3	4	
Hernández et al., 2005	La Higueruela, Toledo, Spain	Olive	None, CC	T, NT, RT, NTM	NT, CC	Rainfed	5	20	2 (4)	3	
Hernanz et al., 2009	El Encín, Madrid, Spain	Wheat-legume rotations	CR	T, RT, NT	RT, NT	Rainfed	20	40	2	4	
Hojati et al., 2006	Lavark, Isfahan, Iran	Maize	None, M, SS		OA, LT	Irrigated	4	15	2 (4)	3	
Kong et al., 2005	LTRAS, Davis, California, USA	Wheat and maize rotations	CR, CC, C		ORG, CC, CMP, Unfert.	Rainfed, irrigated	10	15	6 (10)	3	
Laudicina et al., 2010	Palermo, Siciliy, Italy	Vegetables	C	T, RT	OA, RT, CMP	Irrigated	9	20	3	4	OA applied in CONV
Lithourgidis et al., 2007	Aristotle University, Thessaloniki, Greece	Maize	None, Slurry		Slurry, Unfert.	Irrigated	4	30	2 (4)	6	
López-Bellido et al., 2010	Córdoba, Spain	Wheat rotations	CR	T, NT	NT	Rainfed	20	90	5	12	
López-Garrido et al., 2009	IRNAS-CSIC, Sevilla, Spain	Wheat rotations	CR	T, NT	NT	Rainfed	3	10	1	3	Sample size from Moreno et al., 1997. Long term experiment in Moreno et al., 2006.

Lopez-Piñeiro et al., 2008	Elvas, Portugal	Olive	None, OMW		OA, LT	Rainfed	5	30	2	3	
Lopez-Piñeiro et al., 2010	Elvas, Portugal	Olive	None, OMW		LT	Rainfed	8	25	1 (2)	3	
Madejón et al., 2003	Comarca Costa de Huelva, Spain	Orange	None, C, MSW-C, AIW		OA, LT	Irrigated	3	20	2 (3)	8	
Marinari et al., 2010a	Colle Valle Agrinatura (CVI), Viterbo, Italy	Wheat rotations	None, C		ORG, OA	Rainfed	12	20	1	6	
	La Selva (LSI), Grosseto, Italy	Wheat rotations	None, C		ORG, OA	Rainfed	25	20	1	6	
Marinari et al., 2010b	University of Tuscia, Viterbo, Italy	Wheat-pea-tomato	None, CR, CC, C		ORG, CMP	Irrigated	7	20	1	9	
Martín-Rueda et al., 2007	Alcalá de Henares, Madrid, Spain	Barley rotations	CR	T, RT, NT	RT, NT	Rainfed	4	30	2	4	
Martínez et al., 2008	Antumapu, Chile	Wheat-maize	CR	T, NT	NT	Rainfed/Irrigated	5.5	15	1	3	Final SOC values from Reyes et al., 2002
Matsi et al., 2003	Aristotle University, Thessaloniki, Greece	Wheat	None, Slurry		Slurry, Unfert.	Irrigated	4	30	2 (4)	6	
Mazzoncini et al., 2010	MASCOT, CIRAA, Pisa, Italy	Wheat rotations	CR		ORG, CC	Rainfed/irrigated	5	25	1	3	Detailed C input
Mazzoncini et al., 2011	CIRAA, Pisa, Italy	Maize and wheat rotations	CR, CC	T, NT	NT, CC	Rainfed	15	30	2 (4)	3	
Melero et al., 2006	Torres, Alcalá del Río, Sevilla, Spain	Vegetables (many)	None, C, CR		ORG, CMP	Irrigated	6	15	1	4	
Melero et al., 2007	Tomejil, Carmona, Sevilla, Spain	Wheat rotation	None, M-C, AIW-C		ORG, OA	Rainfed	4	15	1 (2)	4	
Melero et al., 2008	Torres-Tomejil, Alcalá del Río, Sevilla, Spain	Vegetables (many)	None, M-C, AIW-C		ORG, OA	Irrigated	3	15	1 (2)	4	
Melero et al., 2009	IRNAS-CSIC, Sevilla, Spain	Wheat rotation	CR	T	RT	Rainfed	16	20	1	3	Initial C from Ordóñez-Fernández et al., 2007
	Tomejil, Carmona, Sevilla,	Wheat-	CR	T	NT	Rainfed	26	20	1	3	

	Spain	sunflower-legume									
Monokrousos et al., 2006	Kria Vrisi, Thessaloniki, Greece	Asparagus	None, C		ORG, OA	Irrigated	6	10	1 (3)	5	
Montanaro et al., 2009	Basilicata, Italy	Kiwifruit	CR, CC, C	T, NTM	CMP	Irrigated	4	90	1	3	Pruning residues mulched in CMP treatment
	Basilicata, Italy	Apricot	CR, CC, C	T, NTM	CMP	Irrigated	4	90	1	3	Pruning residues mulched in CMP treatment
Montemurro et al., 2010a	Agronomic Research Institute, Foggia, Italy	Lettuce	None, C, AIW, OMW-C		OA, Unfert.	Rainfed?	3	40	2 (5)	3	
Montemurro et al., 2010b	Agostinelli, Rutigliano, Apulia, Italy	Alfalfa and cocksfoot	None, MSW-C, OMW, C		LT	Irrigated	4	40	2 (4)	3	
Moreno et al., 2006	Sevilla, Spain	Wheat-sunflower	None, CR	RT	RT	Rainfed	10	40	1	3	
Morra et al., 2010	Pontecagnano, Salerno, Italy	Tomato-snap bean-lettuce	None, MSW-C		LT, Unfert.	Irrigated	3	30	2 (6)	3	
Moussa-Machraoui et al., 2010	Mahassen, Kef, Tunisia	Wheat-barley	CR	T, NT	NT	Rainfed	4	20	2	4	Seasons used as replicates
	Krib, Siliana, Tuninsia	Wheat-Pea-Oat	CR	T, NT	NT	Rainfed	4	20	3	4	Seasons used as replicates
Mrabet et al., 2001	Sidi El Aydi, Settat, Morocco	Wheat rotations	CR	T, NT	NT	Rainfed	11	20	1	3	
Muñoz et al., 2007	Extremadura, Spain	Maize	None, CR, CC	T, NT	NT, CMP	Irrigated	3, 9	30	2 (3)	4	Stony soil
Nieto et al., 2010	Jaén, Spain	Olive	None, OMW, CR	T, NT	LT	Rainfed	10	30	1	3	
	Jaén, Spain	Olive	None, OMW, CR	T, NT	LT	Rainfed	6	30	1	3	
Okur et al., 2009	Manisa, Turkey	Vineyard	None, M, CC		ORG, CC, CMP	Irrigated	10	20	2 (3)	3	
Pardo et al., 2009	Zaragoza	Wheat-fallow-barley-vetch	CR, CC, C		ORG, OA, Unfert.	Rainfed	6	30	4	9	Coarse fraction quantified

Plaza et al., 2004	La Higueruela, Toledo, Spain	Barley	None, Slurry		Slurry, Unfert.	Rainfed	4	15	2 (6)	3	
Raviv et al., 2006	Israel	Stone fruit	None, CC, C		ORG, CMP	Irrigated	5	30	2	3	
Reganold et al., 2010	Watsonville, California, USA	Strawbery	CC, C		ORG, CMP	Irrigated		30	1	13	
Romanya and Rovira, 2009	Ebro Valley, Spain	Barley-fallow	None, M		ORG	Rainfed	18	30	1	4	
	Ebro Valley, Spain	Wheat and wheat-pea	None, M		ORG	Irrigated	18	30	1	4	
Ryan et al., 2008	Tel Hadya, ICARDA, Aleppo, Syria	Wheat rotations	CR	RT	RT	Rainfed	14	20	1 (7)	3	Initial SOC used as CONV
Sombrero and De Benito, 2010	Torrepadierne, Burgos, Spain	Wheat rotations	None, CR	T, RT, NT	RT, NT	Rainfed	10	30	2	4	% C in residues from Carranca et al., 2009
Sommer et al., 2011	Tel Hadya, ICARDA, Aleppo, Syria	Cereal rotations	None, CR, C	T, RT	RT, OA, CMP	Rainfed	6	30	5 (15)	4	12-year measurements discarded because of lower depth and sample size
Tejada et al., 2006	Sevilla	Wheat	None, AIW, AIW-C		OA	Rainfed	5	25	1 (6)	3	Raw vinasse
Vavoulidou et al., 2006	Santorini, Greece	Vineyard			ORG			30	1	5	Comparisons have been merged because of low sample size
Vavoulidou et al., 2009	Attiki, Mesinia and Arkadia, Peloponnes, Greece	Vineyard, olive			ORG			30	3	4, 6	Sites with n<3 have been discarded
	Chania, Crete, Greece	Olive, citrus			ORG			30	2	4, 3	
Veenstra et al., 2007	Five Points, San Joaquin Valley, California, USA	Cotton-tomato	None, CC	T, RT	CC, RT	Irrigated	5	30	2 (3)	4	
Virto et al., 2007	Olite, Navarre, Spain	Barley	None, CR	T, NT	NT	Rainfed	10	30	1 (2)	4	

[1]CR: crop residues; CC: cover crops; C: compost (when it is preceded by other word, it means that it is made from that material); M: animal manure; MSW: municipal solid wastes; SS: sewage sludge; OMW: olive mill waste; AIW: agro-industrial waste

[2]T: conventional tillage; RT: reduced tillage; NT: no-tillage using herbicides; NTM: no-tillage by mowing.

[3] All data sets include a Conventional (CONV) treatment for comparison. ORG: organic management; LT: land treatment (urban wastes and C inputs exceeding 10 Mg C ha^{-1} yr^{-1}); OA: organic amendments; CC: cover crops; Slurry: liquid manures; NT: no tillage; RT: reduced tillage; CMP: combined management practices (OA combined with CC, CR, RT or NT); Unfertilized: no organic or synthetic fertilizers are applied.

[4]When Duration is absent, C sequestration rate cannot be calculated.

[5]Number of independent data sets used in SOC content meta-analysis. Total number of data sets before aggregation is given in parentheses. See Methods for independence criteria.

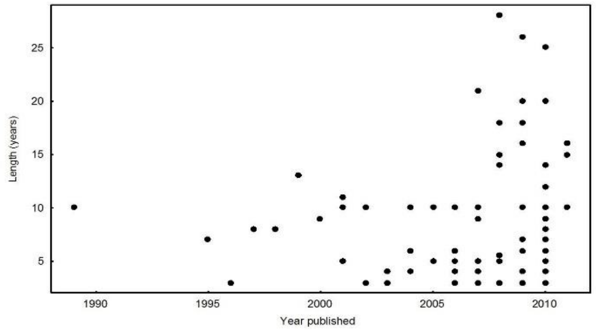

Fig. A.1. Evolution over time of experiment length in the reviewed studies

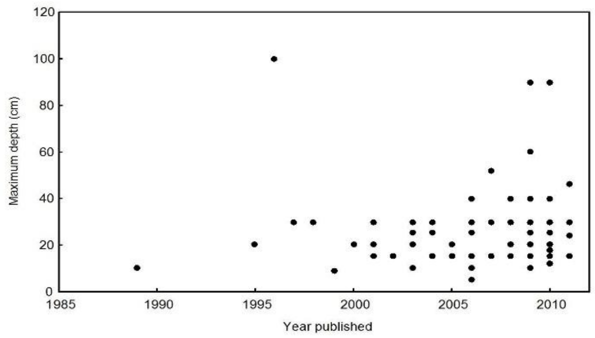

Fig. A.2. Evolution over time of maximum sampling depth in the reviewed studies

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4.3. Study 3.

Greenhouse gas emissions from conventional and organic cropping systems in Spain. I. Herbaceous crops.

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Eduardo Aguilera*, Gloria Guzmán, Antonio Alonso

Agro-ecosystems History Laboratory, Universidad Pablo de Olavide, Carretera de Utrera, km. 1, 41013 Seville, Spain

*Corresponding author: emagufer@upo.es

Keywords

Life cycle assessment; Greenhouse gas emission; Mediterranean; Organic management; Carbon sequestration; Nitrous oxide; Coproduct allocation

Abstract

Agriculture is a major driver of climate change, particularly when all indirect emission sources are accounted for. Mitigation options targeted on one process are often proposed ignoring their secondary effects on the overall greenhouse gas balance. Integrative methodologies such as life cycle assessment (LCA) are often applied without adjusting emission factors to specific site characteristics. Here we used LCA to calculate the global warming potential of 38 pairs of organic and conventional herbaceous cropping systems and products in Spain. Crop products included rainfed cereals and pulses, rice, open-air vegetables and greenhouse vegetables. We used data from farmer interviews and published conversion factors. Our results show that the emission balances were dominated by fossil fuel use rather than by direct field emissions. Organic management reduced crop emissions by 36-65%, with the exception of rice showing an increase of 8% due to methane generation. Product-based emissions of organic crops were also lower by 30% on average, except for rice.

1. Introduction

Agriculture is a major source of greenhouse gases, and is also indirectly responsible for a large share of the greenhouse gases emitted in deforestation and other land use changes. Arable agriculture produces direct and indirect nitrous oxide (N_2O) emissions from nitrogen (N) application to soils, methane (CH_4) from rice paddies, carbon dioxide (CO_2) from direct fossil energy use and N_2O and CH_4 from open biomass burning. Further emissions occur due to fossil energy use in the production of agricultural inputs, particularly N fertilizers. Last, the other major link between agriculture and climate change is soil carbon (C) balance. Historical C losses from agricultural soils have contributed to the increase in both the global greenhouse gas budget and the vulnerability of agriculture to climate change (Lal et al. 2011), but they could be reversed by adequate agricultural practices.

Most mitigation practices can be applied in conventional and organic systems, but broad differences between both management types can be identified. Organic farming aims to reduce the environmental impact of agriculture by avoiding the use of synthetic compounds such as fertilizers and pesticides and by promoting practices such as crop diversification and organic fertilizers. Organic farmers thus prevent fossil energy emission associated to the industrial production of many inputs and promote soil C accumulation. Enhanced SOC under organic management is supported by extensive experimental data in Mediterranean cropping systems (Aguilera et al. 2013a). In spite of this, lower yields could offset reductions in greenhouse gases emissions when quantified on product basis (Nemecek et al. 2011; Venkat 2012). Spain is the country with the largest surface under organic farming in the European Union (EU-25), with 1.65 Mha or 6.5% of total agricultural area (MAGRAMA 2011). The climate in Spain is mostly Mediterranean, with wet, mild winters and hot, dry summers. Herbaceous systems in Spain produce the majority of the local protein and high-value crops for export to nearby regions such as off-season vegetables. Rainfed cereals and legumes are cultivated in vast areas across the country with relatively low yields and low management intensities, occupying 5.36 Mha, or 83% of grain cultivated area, and producing 65% of the grain (MAGRAMA 2011). Vegetables represent one of the major agricultural commodities produced in Spain, with 44% of local production being exported according to FAOSTAT (FAO, 2014). They are mostly cultivated under irrigation, and greenhouse systems represent 18% of the area, with yields usually doubling those of open-air horticulture (MAGRAMA 2011).

A number of papers studying greenhouse gases emissions in Mediterranean herbaceous systems have been published (e.g. Biswas et al. 2008; Venkat, 2012; Theurl et al. 2013), although there is still a lack of information on some specific systems, such as rice paddies, and particularly of comprehensive comparisons between organic and conventional management. On the other hand, most agricultural life cycle assessment (LCA) studies employ Tier 1 IPCC N_2O emission factors and/or do not account for C sequestration, despite soil processes represent a large share of the carbon footprint of agricultural systems and are associated to very high uncertainty (IPCC, 2006). All this may result in inaccurate estimations of total global warming potential, as shown by different studies under Mediterranean conditions (Biswas et al. 2008; Venkat 2012). Another common gap in LCAs of crop products is the study of the effects of the alternative destinies of coproducts such as cereal straw, particularly of their use as animal feed, energy production and soil application for carbon sequestration. This is a very relevant issue in a context of increasing interest for straw energy production (e.g. Cherubini and Ulgiati 2010; Nguyen et al. 2013).

Given the lack of precision of IPCC Tier 1 factors under Mediterranean climate, and the difficulty of implementing intensive field measurements, there is a need to develop simple models able to estimate soil greenhouse gases emissions balances from management information in these areas. We had previously synthetized the current scientific information on climate-specific N_2O emission factors and C sequestration responses to management practices under Mediterranean conditions in two meta-analyses (respectively, Aguilera et al. 2013b, a). In the present work, we combined information from those reviews with data on agricultural management obtained from interviews, conversion factors from LCA databases and the literature and IPCC emission factors to compare for the first time the full greenhouse gases emissions balance, or carbon footprint, of a representative sample of Spanish herbaceous crops under conventional and organic management. We calculated the balances according to LCA procedures, with the following objectives:

1. Determine the influence of organic management on area-based and product-based greenhouse gas balance of a range of herbaceous crops representing organic production in Spain
2. Identify critical processes in the carbon footprint of organic and conventional Mediterranean cropping systems and crop products
3. Analyze the effect of N_2O emission factor choice and coproduct consideration (allocation and system expansion) in the estimation of the carbon footprint of the main product
4. Identify critical options for improving the carbon footprint of organic and conventional Mediterranean cropping systems and crop products

2. Materials and Methods

2.1 Data collection

Information on management and production features was obtained from personal interviews to a representative sample comprised of 38 pairs of organic and conventional farmers in Spain (Alonso and Guzmán 2010). The database was comprised by 18 crop species, grouped by 5 crop types: *rainfed cereals*, including barley (4 pairs of interviews), wheat (3) and oats (1); *rainfed legumes*, including peas (4), fava beans (1) and vetch (1); *rice* (3 pairs); *horticultural open air*, including asparagus(1), lettuce (2), melon (2), celery (1), cauliflower (1), potato (1), broccoli (1), onions (2) and beans (2); and *horticultural greenhouse*, including cherry tomato (1), tomato (4), lettuce (1) pepper (1) and beans (1).

Figure 1. Organic open-air vegetable cultivation in South Spain

2.2 LCA scope

An attributional LCA was performed based on the "cradle to farm-gate" perspective, which considers all inputs and processes for the plant production as well as all the necessary upstream processes. In particular, the following processes were included within system boundaries: manufacturing and maintenance of machinery, fuel production and combustion, greenhouse infrastructure, fertilizers manufacturing, pesticide production, soil N_2O emissions, indirect N_2O emissions,N_2O and CH_4 emissions from open biomass burning, CH_4 emissions from rice paddies, and soil C balance. The temporal boundaries were adjusted to 100 years and all emissions were converted to CO_2 equivalents using IPCC (2006) coefficients. Emissions associated to manure production were excluded from system boundaries, considering that they belong to the animal production systems (Nemecek et al. 2011; Venkat 2012).

Both one hectare of cultivated land and one kg of marketed product were chosen as functional units. Some of the studied cropping systems produced coproducts in addition to the main marketable product, specifically legumes and cereals. These coproducts can be either unused (burned, landfilled), reused (incorporated to the soil) or extracted as additional products for feed or energy valorization (the latter not widespread now). As cereal and legume straw are mainly used for animal production as feed and bedding material, this coproduct was included in the impact assessment through economic allocation. IPCC (2006) Tier 1 methodology was used for the calculation the change in CH_4 emissions resulting from the substitution of cereal grains by straw. Vegetable residues were considered as waste products and no emissions were allocated to them.

2.3 Carbon footprint method

2.3.1 Emissions from the production of inputs

Data on machinery, fuel and electricity consumption and greenhouse infrastructure material requirements were estimated as described in Alonso and Guzmán (2010). According to this procedure, the resource consumption of machinery and implements is attributable to four factors: production of raw materials, manufacture, repair and maintenance and fuel consumption. In this work we included minor changes in the calculation of fuel consumption. In particular, a load ratio of 75% 45% was taken for light duties (atomizing, spraying, bar rolling and hydraulic sweeper). Emissions associated to the production of most inputs from the technosphere, such as machinery, pesticides and some fertilizers were modeled using databases contained in SimaPro 7.2 software (PRé Consultants, 2010), including ecoinvent 2.0 (ecoinvent Centre, 2007) and LCA Food DK (Nielsen et al. 2003) with preference given to ecoinvent 2.0 database. Pesticides were differentiated by compound, family or broad type depending on information availability. Specific Spanish emission factors (Lago et al. 2013), were used for major NPK fertilizers, including ammonium nitrate, ammonium sulphate, ammonium nitrosulphate, urea, N in compound fertilizers, K_2O and P_2O_5. Vermicompost emissions were calculated by scaling compost emissions from ecoinvent by the relative N content. Likewise, the N-scaled ecoinvent emission value for dried poultry manure was taken for all granulated organic fertilizers. N contents of the organic fertilizers were 2.2%, 0.7%, 0.6% and 1.1% for vermicompost, sheep manure, cow manure and chicken manure. Fertilizers were assumed to be transported to the farm by lorry of >16t from distances of 300 km (synthetic fertilizers) and 30 km (organic fertilizers).

Electricity used in irrigation was assumed to come from the 2004 Spanish grid, as modeled in ecoinvent database, and to be half medium voltage and half low voltage. Emission values associated to irrigation infrastructure in the farm were taken from Lal (2004), distinguishing surface irrigation with and without recirculation system, sprinkler and trickle irrigation systems.

Emissions associated to seed production were estimated using the grain emission factors obtained in the present study. This approach implies a self-reference in emission factor calculations, which was solved by iteration of the function until the result stabilized. One emission factor for conventional management and another for organic management were derived for seeds of rainfed cereals, rainfed legumes and rice. In the case of organic management, some of the seed was not of organic origin. This proportion of conventional seed in organic systems was taken into account using Andalusian data from RAS (2006). Andalusia is the largest organic producing region in Spain, representing about half of the certified organic area in the country (MAGRAMA 2011). In order to make the results easier to visualize, all the seed was assumed to be bought in the marked, despite self-production is quite extended in organic farms (RAS 2006). This choice did not influence the results (data not shown). The impacts of vegetable seedling production were estimated based on peat consumption, assuming that the carbon content in peat was 55 kg/m3 (Hall, 2006), that the peat volume used per seedling was 6-40 cm^3, depending on crop species, and that all peat carbon was ultimately released to the atmosphere.

2.3.2 Production and destinies of crop residues

Residual biomass production quantities (straw and other crop residues) were estimated using yield information obtained from the interviews and residue indexes were taken from Guzmán et al. (2014). The only information on residue management available in the interviews referred to the cases in which it was mulched and incorporated to the soil. In the remaining cases, burning rates of 1.2% for cereal straw, 20% for legume straw (fava beans and dry peas), 20% for vegetables and 50% for potatoes were obtained from the National Spanish Emission Inventory (MARM, 2010).

2.3.3 Direct field emissions

N_2O emissions were estimated based on N inputs in the form of organic and synthetic fertilizers and agricultural residues. N content in the residues was obtained from López et al. (2005) (grain crops) and Rahn and Lillywhite (2002) (vegetables). Following IPCC (2006) guidelines, N released from soils with diminishing soil organic C stocks was also accounted as an input for N_2O emission estimation, assuming a C:N ratio in soil organic matter of 10:1. Direct N_2O emissions were calculated using specific Mediterranean factors adjusted to irrigation type (Aguilera et al. 2013a): 0.08%, 0.66% and 1.01% of applied N emitted as N_2O-N for rainfed, drip irrigation and high-water irrigation systems, respectively. These factors are the means of all published information compiled by Aguilera et al. (2013a). IPCC (2006) emission factor of 0.3% was assumed for rice systems. In cereal and vegetable systems, Mediterranean factors were compared through a sensitivity analysis with IPCC factor of 1%. The influence of applying a reduction of 20% in the emission factor of solid organic fertilizers was also studied in the sensitivity analysis, as N_2O emissions associated to organic fertilizers could be lower than those of synthetic ones under Mediterranean conditions (Aguilera et al. 2013a). Indirect emissions were estimated using IPCC (2006) Tier 1 methodology, obtaining a N_2O indirect emission factor of 0.4% N_2O-N per kg of N applied for synthetic fertilizers and 0.5% for organic ones. Emissions of N_2O and CH_4 from biomass burning were calculated following IPCC (2006) Tier 1 methodology.

CH_4 emissions from rice cultivation were estimated following IPCC (2006) Tier 1 guidelines, considering continuous flooding during cultivation and a non-flooding period of <180 days previous to cultivation, and accounting for the amounts of rice straw (applied long before cultivation), weeds and manure applied to the soil.

2.3.4 Carbon sequestration

In Aguilera et al. (Submitted to this issue) we developed a simple model for the calculation sequestration rates based on specific information from Mediterranean cropping systems. Soil C was assumed to be in equilibrium in conventional systems with no organic inputs and conventional tillage, and carbon inputs influenced the balance according to experimental values, which were modified to account for a 100-year time horizon. In this work we adjusted the model to the particularities of herbaceous crops, taking into account the effect of tillage in herbaceous systems, the presence of N inputs in any form and the presence of C inputs in the form of straw and weeds. C content of residues was obtained from Rahn and Lillywhite (2002). We assumed that full tillage was the reference practice, with no associated changes in soil carbon, while no tillage was associated to a net sequestration of 0.15 Mg C per ha and year. This value is the average effect between no tillage and full tillage in the cases with similar C inputs in the meta-analysis by Aguilera et al. (2013b). Using the same source, the absence of some kind of N input in herbaceous systems was associated to a net emission of 0.48 Mg C per ha. We considered that 8.5% of straw C was incorporated into the soil at a 100-year time horizon using data from other studies under Mediterranean conditions (Alvaro-Fuentes and Paustian 2011; Kong et al. 2005). Organic farms usually produce more weed biomass than conventional ones (Guzmán et al. 2014). This difference was considered as an additional C input to organic systems (Table 1).

Table 1. Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) groups of herbaceous crops in Spain. Data refer to one hectare and year unless otherwise stated.

	Rainfed cereals		Rainfed legumes		Rice		Open-air vegetables		Greenhouse vegetables	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Number of interviews	8	8	6	6	3	3	13	13	8	8
Inputs										
Drip irrigation (%)	0%	0%	0%	0%	0%	0%	54%	54%	100%	100%
Surface irrigation (%)	0%	0%	0%	0%	100%	100%	38%	38%	0%	0%
Water use (m^3)	0	0	0	0	11500	11500	3677	3677	3613	3613
Electricity (KWh)	0	0	0	0	3634	3634	1162	1162	1141	1141
Seeds (kg)	183	173	182	192	215	235	0	0	0	0
Seedlings (1000 units)	0.00	0.00	0.00	0.00	0.00	0.00	42.55	40.70	31.90	32.22
Machinery use (h)	8	6	6	5	12	10	27	20	35	36
Fuel consumption (l)	135	109	108	93	211	194	269	219	298	281
Mulching plastic (kg)	0	0	0	0	0	0	35	11	185	168
Mineral nitrogen (kg N)	73	0	13	0	152	0	104	0	264	0
Mineral phosphorus (kg P_2O_5)	38	0	21	0	19	0	54	0	190	0
Mineral potassium (kg K_2O)	34	0	23	0	19	0	141	3	292	0
Manure (Mg)	0.00	1.43	0.00	0.72	0.00	7.67	5.08	18.08	7.00	13.75
Slurry (Mg)	3.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0	0	0	0	0	0	1	6	63	1103
Total carbon inputs (kg)	108	577	407	532	2233	3171	690	3182	1546	2736
Total nitrogen inputs (kg)	87	17	61	50	177	90	162	167	393	155
Synthetic pesticides (kg active matter)	0.43	0.00	0.91	0.00	7.14	0.00	5.46	0.00	52.36	0.00
Sulphur (kg)	1.56	0.00	0.00	0.00	0.00	0.00	2.31	0.23	3.00	42.78
Copper (kg)	0.00	0.00	0.00	0.00	0.33	0.00	3.02	0.20	3.45	2.15
Natural pesticides (kg)	0.00	0.00	0.00	0.00	0.23	0.31	0.00	3.95	0.00	29.44
Steel (kg)	0	0	0	0	0	0	0	0	411	411
Greenhouse cover plastic (kg)	0	0	0	0	0	0	0	0	1208	1208
Concrete (kg)	0	0	0	0	0	0	0	0	7290	7290
Production										
Yield (Mg fresh matter, mean)	2.95	1.95	2.32	1.32	7.50	5.17	23.37	17.48	66.23	47.96
Yield (Mg fresh matter, standard deviation)	1.57	0.62	0.92	0.65	0.50	0.76	16.31	14.67	29.71	17.42
Yield (Mg dry matter, mean)	2.54	1.68	1.99	1.13	6.48	4.46	2.48	1.93	3.66	2.75
Weeds (Mg dry matter)	0.39	0.67	0.39	0.67	0.30	0.64	0.00	2.00	0.00	2.00
Residue yield (Mg dry matter)	3.26	2.15	3.23	1.81	8.19	5.64	3.08	2.32	9.55	7.28
Residue destiny										
Open burning (Mg)	0.04	0.02	0.45	0.21	2.73	0.00	1.11	0.30	1.35	1.27
Soil incorporation (Mg)	0.19	0.73	0.99	0.76	5.46	5.64	0.65	1.16	2.81	0.93
Coproduct (Mg)	3.03	1.40	1.79	0.83	0.00	0.00	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	13%	9%	8%	8%	0%	0%	0%	0%	0%	0%
Soil emissions										
Direct nitrous oxide (kg N_2O)	0.11	0.02	0.03	0.02	0.84	0.42	1.84	1.98	4.03	1.56
Indirect nitrous oxide (kg N_2O)	0.57	0.14	0.18	0.12	1.15	0.71	1.07	1.27	2.64	1.18
Methane (kg CH_4)	0.10	0.05	1.21	0.56	347.60	440.60	3.00	0.81	3.64	3.43
Carbon (kg C)	-60	-55	-69	-100	-381	-633	-180	-740	-352	-654

2.4 Sensitivity analysis

We performed sensitivity analyses of the effects of using different N_2O emission factors in cereals and open-air vegetables and of applying different methods of allocation and system expansion for coproduct consideration in cereals. Nitrous oxide scenarios are compared to *Base*, which represents base case sequestration using Mediterranean factors, and include a scenario based on IPCC Tier 1 methodology (*IPCC*) and another with a 20% reduction in the emission factor of organic fertilizers (*Reduction*).

In a sensitivity analysis of cereal grain production, we compared economic allocation applied in the base case with different methods for coproduct consideration, including allocation and system expansion methods. Coproduct scenarios include the base case scenario (*Economic*); product allocation (*Product*), in which all emissions are allocated to the main product; dry matter allocation (*Dry matter*); and system expansion, in which co-produced straw is assumed to substitute cereal grain for cattle feeding. In *Expansion 1*, emissions from replaced grain production, calculated by the metabolizable energy content of each feedstock, are subtracted from product emissions. In *Expansion 2* the change in enteric CH_4 emissions due to the increase in straw in ruminants diet is also considered, following IPCC (2006) methodology based on total energy intake and feed digestibility.

3. Results and discussion

3.1 The greenhouse gas profiles of the studied systems

The studied categories are composed by heterogeneous cases, including different crop types, study sites and management characteristics. This is reflected in the high variability of yields and carbon footprints that can be observed in Tables 1 and 2. Therefore, the results herein presented must be taken with care. Nonetheless, certain clear trends can be identified between categories, and the pair-wise selection of organic and conventional study cases reduces the influence of external factors on the estimated carbon footprints.

3.1.1 Rainfed grains

Because of the low N_2O emission factor in rainfed Mediterranean cropping systems, the role of this gas was very limited in rainfed cereals, while the production and use of industrial inputs account for a major share of the emission profile. These inputs were mainly comprised of machinery and fuel in the case of organic management, with a higher contribution of fertilizers in the case of conventional management (Fig. 2a). Interestingly, average C sequestration was not higher under organic than under conventional management, despite greater average C inputs, mainly in the form of crop residues. This could happen because there was one organic case with no organic inputs, and subsequently with a high C loss from the soil, according to our model. The net area-based global warming potential was 1024 kg CO_2e per ha of conventional cereal, in comparison with 361 kg per ha of organic cereal (Table 2), with an average decrease of 65% for organic products. The change was reduced to -42% on a product basis due to lower yields under organic management, with emission values of 318 and 185 g CO2e per kg conventional and organic product, respectively. The production of fertilizers was the main factor responsible for the differences observed, while C sequestration was similar between both types of management on an area basis. Our results for conventional grain production are in the lower range of global estimations by Nemecek et al. (2012) and in accordance with studies under Mediterranean climate such as that of Biswas et al. (2008).

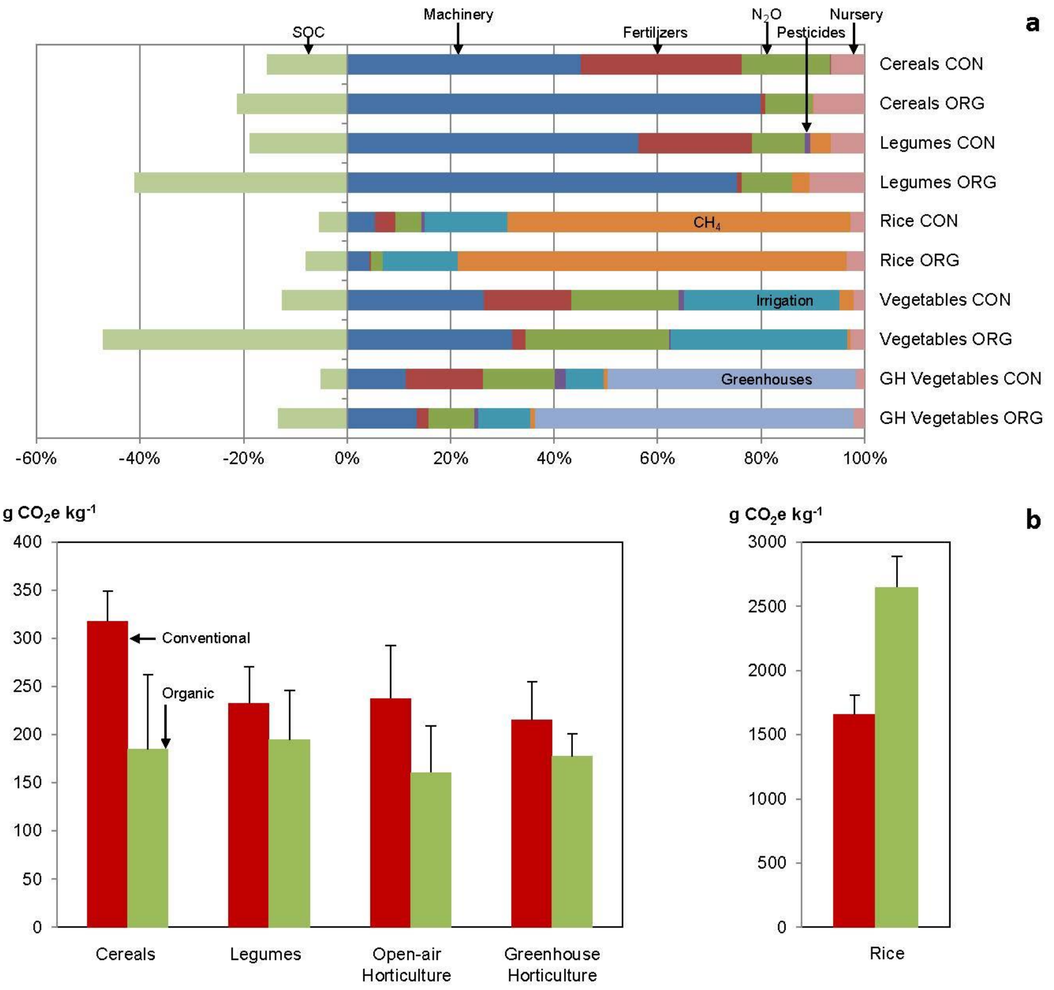

Figure 2. Global warming potential (g CO_2e kg^{-1}) of the five types of crop products and the two types of management (Conventional.*Con*. and Organic.*Org*) studied, expressed as the breakdown of the main processes implicated and as the net balance resulting from subtracting carbon sequestration to total emissions (means with standard errors). The components of the emission balance comprise *Nursery,* including seed production in grain crops and plant nursery in vegetables; *CH_4*, methane from rice cultivation and biomass burning; *Pesticide* production; *N_2O*, including soil emissions, indirect emissions and biomass burning emissions of nitrous oxide; *Fertilizers* production and transport; *Machinery* production and use, including fuel production and use; and *SOC*, which accounts for the changes in soil organic carbon resulting from management practices.

Table 2. Global warming potential of organic and conventional Spanish herbaceous cropping systems for a 100-year time horizon expressed as kilograms of CO_2eper hectare and year and as grams of CO_2e per kg of product.

	Rainfed cereals		Rainfed legumes		Rice		Open-air vegetables		Greenhouse vegetables	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e ha^{-1})										
Machinery production	28	22	21	19	45	42	52	43	64	56
Fuel production	58	47	47	40	91	84	116	95	129	122
Fuel use	364	294	290	251	569	523	725	591	804	758
Fertilizer production	410	5	183	3	521	29	792	73	2364	216
Direct nitrous oxide	33	6	10	6	249	126	547	591	1202	465
Indirect nitrous oxide	170	41	54	35	344	210	318	378	787	352
Pesticides	3	0	9	0	81	3	58	12	362	91
Irrigation infraestructure	0	0	0	0	90	90	206	206	311	311
Irrigation energy	0	0	0	0	2018	2018	645	645	634	634
Methane	2	1	30	14	8690	11015	75	20	91	86
Greenhouse	0	0	0	0	0	0	0	0	5157	5157
Nursery	65	46	42	44	343	501	146	135	164	166
Carbon	-110	-102	-127	-184	-699	-1160	-330	-1399	-645	-1199
Total (mean)	1024	361	568	232	12401	13481	3448	1418	11841	7592
Total (standard deviation)	432	394	287	157	1751	208	1633	557	5216	1150
Product-based emissions (g CO_2e kg^{-1})										
Machinery production	10	11	9	15	6	8	4	5	1	1
Fuel production	22	24	21	32	12	17	9	12	2	3
Fuel use	137	152	131	202	76	103	55	77	14	17
Fertilizer production	117	2	63	3	70	6	46	7	34	5
Direct nitrous oxide	10	3	4	4	33	26	31	47	19	10
Indirect nitrous oxide	54	19	22	25	46	43	23	37	12	8
Pesticides	1	0	3	0	11	1	3	1	5	2
Irrigation infraestructure	0	0	0	0	12	18	23	31	7	8
Irrigation energy	0	0	0	0	270	396	58	73	10	13
Methane	1	0	11	11	1161	2164	7	2	2	2
Greenhouse	0	0	0	0	0	0	0	0	109	126
Nursery	21	21	19	35	47	91	5	8	3	4
Carbon	-58	-50	-54	-135	-93	-228	-34	-144	-12	-27
Total (mean)	315	183	233	195	1660	2644	238	161	215	178
Total (standard deviation)	88	219	94	125	258	415	196	176	113	66
Coproduct	50	27	21	23	0	0	0	0	0	0

Legumes are usually cultivated in rotation with cereals, and their management is similar to theirs in very aspects, particularly machinery use (Table 1). The low use of synthetic inputs under conventional legume management makes it relatively similar to organic legume management in terms of the composition and net value of the greenhouse gas emission balance (Fig. 2), with a modest change of -16% in CO_2e emissions per kg under organic management. Emission values per kg of conventional and organic legumes, of 233 and 195 g CO_2e, respectively, are 60-80% lower than those of different legumes reported in ecoinvent for different European sites (Nemecek et al. 2007). These low emission values support the extended use of legumes in Mediterranean arable crop rotations, adding to their key role as N-fixers and protein suppliers.

3.1.2 Rice

Rice emissions are dominated by methane, with a significant contribution of irrigation (Fig. 2a). The relative performance of organic management in rice systems was the poorest among all crop types considered, mainly due to high methane emissions and low yields. Area-based global warming potential was 9% higher, but the differences were more marked on a product basis, with an average increase of 60% (1658 and 2650 g CO_2e per kg of conventional and organic rice, respectively). Increased methane emissions in organic systems were associated to the incorporation of rice straw and manures. As shown in other studies, C sequestration promoted by these inputs could not overcome the increase in methane emissions in terms of global warming potential (Wang et al. 2012). Straw and manure have a high content of easily decomposable C, which is associated to methane emissions in rice paddies and therefore to a high scaling factor according to IPCC (2006) methodology. Conventional rice emissions are in the lower range of the global estimations reported by Nemecek et al. (2012), while organic rice emissions are in the upper range.

3.1.3 Vegetables

Irrigation in Mediterranean vegetable cropping systems implies high energy consumption and associated emissions for water pumping and infrastructure building (Table 1), but also allows for higher response to, and thus higher use of, fertilizers and pesticides, whose importance in the greenhouse gases emissions balance grows at the expense of machinery emissions (Fig. 2a). The observed decrease in emissions per hectare in organic vegetable cropping systems (-59%

change on average) was due to lower emissions associated to fuel use, fertilizer and pesticide production and a nearly three-fold increase in C sequestration (Table 2). When studied per kg of product, organic emissions were still lower in 12 out of the 13 cases studied, with an average change of -32% (238 and 161 g CO_2e per kg conventional and organic product, respectively). The reduction occurred despite increased N_2O emissions per hectare due to higher indirect N_2O emission factor for organic fertilizers. This could happen because fertilizer intensification in the studied organic systems promotes soil C sequestration through the increase in C inputs, and maintains low emission levels in the production of fertilizers, as it mainly employs organic waste materials. The observed global warming potential for conventional systems agrees with assessments performed in many other conditions (González et al. 2011; Nemecek et al. 2012) including a study under Mediterranean climate in California (Venkat 2012). Therefore, differences with other climates seem to be less marked for irrigated Mediterranean systems than for rainfed ones.

Emissions in greenhouse cropping systems were dominated by greenhouse infrastructure (41% and 59% of emissions in conventional and organic systems, as average), as was also observed in the energy balance of the same crops (Alonso and Guzmán 2010). Emissions in greenhouse cropping systems were relatively high per hectare and low per kg of product in both types of management, due to input intensification and high yields. We observed that the average differences in the net global warming potential between organic and conventional management were relatively small (-17% on a product basis), which can be attributed to the high burden of greenhouse infrastructure and water consumption in these systems. The average emission values of 215 and 178 g CO_2e per kg of conventional and organic vegetables, respectively, confirm the relatively low carbon footprint of vegetable production in Mediterranean greenhouses obtained in other studies, as compared to vegetables cultivated in heated, glass built greenhouses in colder regions (Theurl et al. 2013; González et al. 2011). We found no previous data reporting emissions of organic vegetables cultivated in Mediterranean greenhouses. .

Quality differences should be taken into account when comparing organic and conventional vegetables, as higher dry matter content of organic products (Lester and Saftner, 2011) may partially offset lower yields.

3.2 Sensitivity analysis

3.2.1 Nitrous oxide emission factor

The results showed a high variability in the response to the choice of N_2O emission factor. The use of a reduced emission factor for organic fertilizers had an almost negligible effect on the global warming potential of the studied systems. By contrast, largest changes occurred when IPCC factor was used in conventional cereals, which increased net emissions by 33%, as compared to only 18% in organic cereals (Fig. 3a). N_2O emission in cereals were actually highly affected by the emission factor due to the large difference between Mediterranean rainfed factor and IPCC factor (one order of magnitude lower for Mediterranean), but in organic systems the relative contribution of nitrous oxide to the total carbon footprint was small due to low rates of fertilizer application (Fig. 2a). In vegetables, the net effect of the change of N_2O emission factor was much lower (Fig. 3b) because the Mediterranean factors for irrigated systems used in the base case were very similar to the IPCC factor (they were equal in the case of high-water irrigation and 30% lower for drip irrigation). Our results show that the use of climate-specific emission factors instead of IPCC Tier 1 factors for the estimation of N_2O emissions in LCA of Mediterranean cropping systems leads to substantial differences in the net global warming potential, especially under rainfed conditions. These results agree with the LCA of rainfed cereal systems in Mediterranean-climate Western Australia performed by Biswas et al. (2008).

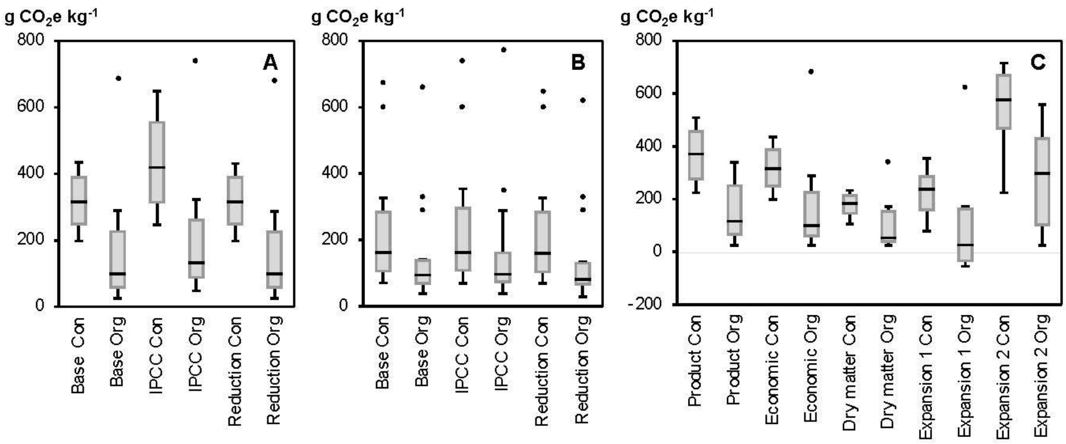

Figure 3. Sensitivity analysis of the global warming potential (g CO_2e kg^{-1}) of conventional (*Con*) and organic (*Org*) management as affected by different estimations of nitrous oxide emissions in cereals (a) and open-air vegetables (b) and as affected by changes in coproduct consideration in

cereals (*c*). Nitrous oxide scenarios are compared to *Base*, which represents base case sequestration using Mediterranean factors and include estimates using IPCC Tier 1 methodology (*IPCC*) and a 20% reduction in the emission factor of organic fertilizers (*Reduction*). Coproduct scenarios include product allocation (*Product*), in which all emissions are allocated to the main product; economic allocation (*Economic*); dry matter allocation (*Dry matter*); and system expansion, in which co-produced straw is assumed to substitute cereal grain for cattle feeding. In *Expansion 1*, emissions from replaced grain production are subtracted from product emissions, and in *Expansion 2* the change in enteric CH_4 emissions is also considered. Data are presented by medians (lines), 25-75% percentiles (boxes), non-outlier ranges (whiskers), outliers (dots) and extremes (asterisks).

3.2.2 Coproduct consideration

The sensitivity test shown in Fig. 3C clearly indicates the relevance of the consideration of this coproduct in the greenhouse gases emissions balance of cereal grains. The different methods compared address different research questions and lead to very different results. The results obtained using economic criteria for allocation (Economic), as in the base scenario, only slightly differed from those obtained allocating all emissions to the main product (Product), due to the low economic value of straw. When mass criteria were used (Dry matter), however, major changes in the estimated carbon footprint were observed, as the production of commercialized straw, in terms of dry matter, was similar to that of grain (Table 1). The largest effect of coproduct consideration was observed when the system was expanded. In Expansion 1 scenario, we subtracted the emissions associated to the production of the grains replaced by straw as animal feed. This resulted in the lowest global warming potential among the studied methodologies. Other studies have also shown that consequential LCAs usually yield lower global warming potential estimates than attributional LCAs (Thomassen et al., 2008). The observed reduction, however, was more than offset by increased enteric CH_4 emissions due to the use of a lower quality feed (Expansion 2). These results highlight the need for a full accounting of the effect of these destinies when studying the potential of straw for C sequestration or for other uses such as energy production. Our analysis suggests that complementing the attributional assessment of agricultural products with a consequential analysis would provide insights for understanding these tradeoffs.

3.3 The potential for mitigation in herbaceous systems

As described in section 3.1, most of the studied agricultural systems show relatively low emission levels, especially under organic management. The high variability observed, however, together with the limited extension of some climate-friendly practices, suggest that the potential for mitigation is still very large. A singular case is the high emission observed in organic rice production, whose reduction would probably benefit from the use of more stabilized organic inputs, such as compost, vermicompost or digested slurries, as well as from the increase in yield performance and the reduction in flooding period and in non-renewable input use.

3.3.1 Reducing fossil fuel-based inputs

Fossil energy use dominated the greenhouse gas profiles of the studied systems. In rainfed systems, most emissions were produced by machinery and fuel use, plus fertilizer production in the case of conventional systems. In irrigated systems, the emission profile was diversified by water extraction and greenhouse infrastructure, which are also based on fossil fuels. Fossil fuel-based inputs can be reduced through efficiency gains and through substitution by self-produced renewables ones. The first strategy could make use of techniques such as drip irrigation and reduced tillage practices. The second would imply the use of renewable energy in irrigation, such as solar or wind energy, and the self-production of the fuel used in the farm. Drip irrigation clearly appears as a win-win strategy, saving water and the energy needed for its extraction, while potentially lowering N_2O emissions (Aguilera et al. 2013a). Our results show that the increased emissions due to drip irrigation infrastructure are clearly offset by the emission savings.

In mechanization-dominated rainfed systems, reduced tillage would promote C sequestration (Aguilera et al. 2013b) while saving fuel, cutting greenhouse gases emissions and reducing the dependence on the increasingly scarce supply of oil derivatives. Tillage reduction often relies on chemical weed control, that may increase pesticide production emissions, but purely mechanical minimum tillage methods are also available. Fuel savings would also improve the feasibility of selfproducing the fuel needed in the farm, as a smaller fraction of the product would be needed for producing the biofuel. Self-production of the fuel could increase the efficiency in the use of non-renewable energy while producing a protein-rich coproduct (Aguilera 2009).

3.3.2 Maximizing total yield

Lower yields in organic systems were responsible for decreased carbon footprint reductions under organic farming when studied on a product basis instead of on an area basis. Increasing yields thus appears as a priority for the improvement of the environmental profile of organic cropping systems. Yield improvement would require solving problems associated to the management of pests, diseases, weeds and nutrients. On the other hand, proposals for changes in management practices have to consider their full impacts. For example, weeds also contribute to the reduction of the net global warming potential by promoting carbon sequestration, as well as to pest control by the provision of habitats for biological control agents. Furthermore, in forage-oriented systems such as cereal and legume fields, the production of weeds may represent an additional forage output that could increase total forage yield when compared with weed free monocultures (Gholamhoseini et al. 2013). Therefore, deriving weed management recommendations in relation with climate change mitigation is not straightforward and should take into account the multiple functions of weeds in cropping systems. A similar problem appears with the valorization of straw as an additional product. The direct utilization of straw for electricity or thermal energy production is gaining increasing interest (Nguyen et al. 2013), but our analysis suggests that the extraction of straw for energetic purposes should always consider the effect on other uses such as animal feeding and the needed straw to be retained for maintaining or increasing SOC balance. Hence, the revision of yield metrics to consider the multifunctionality of the primary production of cropping systems would contribute to a more accurate quantification of their yield-related environmental burdens, helping to reduce the land cost of sustainability (Guzmán et al. 2011). From this view, techniques that simultaneously address multiple functions seem the most interesting.

3.3.3 Increasing carbon sequestration

Our data suggest that enhancing C sequestration leads to climate change mitigation in Mediterranean systems, which usually show low N_2O emissions and a high soil C response to organic inputs. On the other hand, our estimation of carbon sequestration is based on a 100-year time horizon. Higher carbon sequestration rates, and thus a higher influence of carbon sequestration in the overall balance, would be occurring at a shorter time frames (i.e. 20 years), as shown in a sensitivity analysis of the carbon footprint of Spanish fruit tree orchards (Aguilera et al. Submitted to this issue).

Maximizing organic matter inputs both in organic and conventional systems is a way to achieve carbon sequestration that could help improving yields in the former, while reducing the need for synthetic fertilizers and their associated emissions in the latter. However, a number of factors limit endogenous-based C sequestration in herbaceous systems. A major structural limit is space availability for cultivating cover crops, as it would usually mean cultivating less commercial crops in the rotations. The majority of the potential for expanding cover crops relies in crop rotations including a year of fallow and in summer irrigated crops in which the soil is now left bare in the winter.

Another limitation to the increase in organic inputs for soil C sequestration is the alternative use of straws as animal feed. In a country such as Spain, highly dependent on feed imports (Lassaletta et al. 2014), straw may be playing an important role preventing the use of imported grains. Our calculations show that this replacement would have a larger effect on the carbon footprint of crop products than its application to soil. At the same time, increasing soil organic matter levels in Mediterranean soils may be key for adapting to climate change, due to its positive effect on soil physical properties (Aguilera et al. 2013b). This tradeoff suggests that there is a need to ensure that the majority of the C removed with straw returns to the soil in some form. The transformation of plant biomass in ruminant bodies implies a release of C as CO_2 and CH_4, but could also contribute to stabilization of the C remaining in the manure, potentially resulting in similar soil retention of the original C with manures than with raw plant biomass (Thomsen et al. 2013). Accordingly, some studies have shown that grazing cereal stubble may not have detrimental effects on C stocks, at least under semi-arid conditions (Quiroga et al. 2009). In spite of this potential, however, the comparison of the global warming potential of these alternative uses of straw would also require considering CH_4 and N_2O emissions produced by animal raising and the potential of dietary changes for the reduction of the demand of meat.

4. Conclusions

Our analysis of the greenhouse gas emission profile of 38 pairs of conventional and organic herbaceous cropping systems in Spain shows that energy-related emissions are the main contributors to the net global warming potential of most of the studied systems, followed by C sequestration. The other soil emissions usually represent a minor role, especially in rainfed systems, which are dominated by fuel emissions. Relatively low N_2O emissions under Mediterranean conditions, due to low N application rates and low N_2O emission factors as compared to global IPCC factors (2006), are partially responsible for this. These data suggest that management recommendations for climate change mitigation should be based on comprehensive approaches to the quantification of agricultural greenhouse gases, including upstream life cycle processes and climate-specific calculations of direct field emissions and carbon sequestration.

Despite a high variability, we observed a general trend for lower greenhouse gases emissions under organic management, with the significant exception of rice, which represents a clear outlier in our analysis. Higher methane emissions under organic farming point at the convenience of using more stabilized organic inputs in organic rice systems. In the other systems, emission savings under organic farming were due to lower input use or higher C sequestration, or to the combination of both processes, and they were not fully offset by lower yields.

Our results show for the first time that Mediterranean conditions favor intensification in the application of organic matter inputs for climate change mitigation in rainfed and drip-irrigated systems, because the higher C sequestration and yields achieved are not accompanied by very high increases in N_2O emissions. On the contrary, this effect is jeopardized by high fossil energy consumption in surface-irrigated systems and greenhouse systems. More research is needed to verify if similar patterns are found in other semi-arid climates. The dominance of non-renewable energy emissions in the global warming potential of the studied crop products suggests that strategies aiming to reduce resource consumption would successfully contribute to climate change mitigation. For instance, drip irrigation increases yield and residual biomass with low water use and N_2O emissions. Non-renewable fuels could be substituted by self-produced ones, and irrigation water could be extracted with solar or wind energy.

Finally, our results underline the importance of the current and potential multifunctionality of coproducts in the greenhouse gases emissions balance of crop products. For example, the use of herbaceous residues and weeds as animal feed, ensuring that the manure is appropriately applied to the soil, could produce extra food with minor effects on soil C balance, but also promotes CH_4 emissions. Thus, this practice seems to be more resource efficient but may increase net greenhouse gases emissions as compared to directly incorporating those residues into the soil. In any case, the role of straw and weeds as ruminant feed needs to be taken into account in the assessment of alternative uses such as C sequestration or energy provision.

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4.4. Study 4.

Greenhouse gas emissions from conventional and organic cropping systems in Spain. II. Fruit tree orchards

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Eduardo Aguilera*, Gloria Guzmán, Antonio Alonso

Agro-ecosystems History Laboratory, Universidad Pablo de Olavide, Carretera de Utrera, km. 1, 41013 Seville, Spain

*Corresponding author: emagufer@upo.es

Keywords

Life cycle assessment; Mediterranean; Orchards; Organic management; Carbon sequestration; Cover crops; Prunings; Coproduct allocation

Abstract

Fruit tree orchards have an historical and economic importance for Mediterranean agriculture, notably in Spain. Fruit tree orchards have the potential to mitigate global warming by sequestrating carbon (C) and providing renewable fuels. Actually there is few information on the benefits of organic practices. Therefore we analyzed the greenhouse gas contribution of 42 pairs of organic and conventional perennial cropping systems, including citrus, subtropical trees, other fruit trees, treenuts, vineyards and olives, using life cycle assessment (LCA). The assessment was based on management information from interviews and involved the estimation of soil carbon sequestration, specific Mediterranean N_2O emission factors and the consideration of coproducts. Results show on average a 56% decrease of greenhouse gas emissions under organic versus conventional cropping, on an area basis. On a product basis greenhouse gas emissions decreased by 39% on average. These findings are explained mainly by C

sequestration in soils, which is due in turn to higher C inputs by cover cropping and incorporation of pruning residues.

1. Introduction

In an energy-constrained and greenhouse gases-saturated world, woody perennial cropping systems show numerous advantages. Their major differentiating feature is biomass accumulation in living tissues during crop growth, which leads to net carbon accumulation when fruit tree orchards substitute herbaceous crops (IPCC 2006). This biomass can be harvested when plantations are renewed, and then burned in substitution of fossil fuels or temporally sequestered as wood products. Fruit tree orchards also produce large amounts of residual biomass in the form pruning residues, which can be used for soil conditioning, animal feeding, or for energetic purposes (e.g. Infante-Amate and González de Molina 2013; Kroodsma and Field 2006). Fruit tree orchards were estimated to supply ca. 80% of total fuelwood consumed in Spain in the year 2000 (Infante-Amate et al. 2014). In addition, cover crops can be established below the trees protecting the soil from erosion, contributing to soil carbon sequestration (González-Sánchez et al. 2012; Aguilera et al. 2013a) and potentially serving as animal feed (Ramos et al. 2011). In spite of these promising features, there is some concern about the temporal limitation of soil carbon sequestration, which would only occur until a new equilibrium is reached. For example, González-Sánchez et al. (2012) observed that C sequestration rate under cover crops in Spanish fruit tree orchards was 1.59 Mg C/ha in short-term (<10 years) experiments and 0.35 Mg C/ha in long term (>10 years) experiments. On the other hand, tree cropping systems following forest may induce a decrease in soil organic carbon, at least during the establishment period (Noponen et al. 2013), all of which suggests that there is a need to optimize the management in order to maximize soil quality and carbon content in orchards.

Woody perennial crops usually dominate the landscape and the rural economy in producing areas of the Mediterranean region. They include many species originally domesticated in this climatic area, and sometimes biophysically limited to it. The specialization of olive, grape and citrus production was largely responsible for the agricultural modernization in the Mediterranean, driven by the growing demand from an expanding world market of these commodities, but they were made possible by the multifunctional character of Mediterranean woody systems (Infante-Amate and González de Molina 2013). The relevance of fruit tree orchards has grown further in the last decades with the expansion of new commercial crops such as pip and stone fruits, subtropical fruits and some treenuts. The expansion of perennial woody systems such as vineyards and orchards in California has been associated to the accumulation of significant amounts of carbon in their living biomass (Kroodsma and Field 2006). Also in Andalucía, in South Spain, an increase in vegetation carbon stocks between 1956 and 2007 was partially associated to the expansion of permanent crops (Muñoz-Rojas et al. 2011).

Fruit tree orchards also supply a relevant fraction of the Spanish diet, representing nearly 20% of dietary energy intake according to FAOSTAT (FAO, 2014), mainly in the form of olive oil. Nowadays almost one third of total cropland and about one half of organic cropland in Spain are cropped to fruit tree orchards (MAGRAMA 2011). The high relative importance of woody systems under organic farming is probably influenced by the availability of self-produced organic matter sources allowing to better close nutrient cycles. Specific features of organic farming could further improve the contribution of fruit tree orchards to climate change mitigation. Reduced fossil energy consumption in organic orchards (Guzmán and Alonso 2008; Alonso and Guzmán 2010) suggest that there may also exist greenhouse gases emissions savings in these systems, taking into account the high importance of energy-related emissions in the carbon footprint of herbaceous Spanish cropping systems (Aguilera et al. Submitted to this issue). At the soil level, evidence suggests low N_2O emissions associated to organic fertilizers (Aguilera et al. 2013b), and higher SOC stocks in organically managed soils under Mediterranean climate (Aguilera et al. 2013a). All these features point at a large mitigation potential of Mediterranean fruit tree orchards, and particularly of organic agroecosystems, but comprehensive assessments of their global warming potential are very scarce.

In the present work we applied LCA methodology to analyze the full greenhouse gases emissions balance of the most relevant perennial cropping systems in Spain and compare the performance of organic and conventional management. The analysis included all emissions involved at the production step, incorporating specific N_2O emission factors and estimations of carbon sequestration and fuel wood coproduction. We calculated the balances following LCA procedures, with the following objectives:

1. Determine the influence of organic management on area-based and product-based greenhouse gases emissions in a range of fruit tree orchards representing organic production in Spain
2. Identify critical processes implied in the global warming potential of organic and conventional Mediterranean fruit tree orchard products.
3. Analyze the effect of carbon sequestration rate calculation and coproduct consideration on the total global warming potential.
4. Identify critical options for improving the carbon footprint of organic and conventional Mediterranean fruit tree orchard products

2. Methods

2.1 Data collection

Information on management and production features was obtained from personal interviews to a representative sample comprised of 42 pairs of organic and conventional farmers in Spain, as described in Alonso and Guzmán (2010). Two cases were added to this set of interviews, one for vineyards and one for carob tree. In addition, to account for the large change occurred in olive organic management in the last five years, four olive cases (4, 5, 6, 7) were substituted for more recent interviews, performed in 2012. The complete database comprised 16 crop species, grouped by 6 crop types: *citrus*, including mandarins (2 pairs of interviews) and oranges (3); *fruits*, including apples (4), pears (2), plumb (1), table grapes (1), peach (1), apricot (1) and figs (2); *subtropical fruits*, including avocado (2), mango (1) and bananas (2); *treenuts*, including almonds (3), hazelnuts (2) and carob tree (1); *vineyards*, including grapes for wine (7); and *olives* (7 pairs). Banana was included in the subtropical fruit category despite being an herbaceous crop because of its similarities with tree orchard cropping systems.

Figure 1. Conventional (left) and organic (right) olive orchards in South Spain

2.2 LCA scope

An attributional LCA was performed based on the "cradle to farm-gate" perspective, which considers all inputs and processes for the plant production as well as all the necessary upstream processes (see details in Aguilera et al. Submitted to this issue). The temporal boundaries were adjusted to 100 years as recommended by IPCC (2006). This extends the study to the full life cycles of the studied crops, including the unproductive years of perennial cropping systems and long-term soil carbon dynamics. Unproductive period of perennial cropping systems were estimated according to cropping cycles data from Fernández-Escobar (1988). Tree crop plantations cycles were divided in three periods: i) Implantation period, with no yield and 50% fertilizer rate; ii) Growing period, with 50% yield and 50% fertilizer rate; and iii) Full-production period, with 100% yield and 100% fertilizer rate. Net emissions in the implantation and growing periods were computed as “Unproductive stages” in the impact assessment.

Carbon accumulation in the living biomass was not considered a carbon sink, as plantation surfaces were assumed to be stable in time. However, this biomass is commonly used as fuelwood after plantation removal, so its annual accumulation was quantified. In the same way, pruning residues are usually burned in the field or more recently incorporated to the soil, but a non-negligible share (the thick branches) is commonly used as fuelwood (Infante-Amate and González de Molina 2013). Economic allocation was applied to total fuelwood coproduction (removals and prunings) in the main analysis. Economic value of the crop products and the fuelwood were obtained mainly from CAPMA (2013) and MAGRAMA (2009).

2.2 Carbon footprint method

2.3.1 Emissions from the production of inputs

Data on material and energy consumption associated to the agricultural operations were estimated according to Alonso and Guzmán (2010) with modifications described in Aguilera et al. (Submitted to this issue). Emissions factors for the production of inputs from the technosphere were mainly obtained from databases in SimaPro 7.2 software (PRè Consultants 2010), as described in Aguilera et al. (Submitted to this issue). In the case of seeded cover crops, emissions values for the production of legume and cereal seeds were taken from Aguilera et al. (op. cit.).

2.3.2 Production and destinies of crop residues

Pruning residue dry matter production was calculated using residue indexes from Guzmán et al. (2014). The only information on residue management available in the interviews referred to the cases in which it was mulched and incorporated to the soil. In the remaining cases, we assumed that 78% of the prunings were burned (MARM 2008), and 22% represented a coproduct. The same fraction was assumed to be a coproduct in the cases in which the residue was mulched. Living biomass accumulation in fruit tree orchards was estimated from pruning values using a relationship of 0.68 kg accumulated dry matter per kg pruned dry matter (Roccuzzo et al. 2012). This value was roughly validated through interviews to companies in the sector.

2.3.4 Direct field emissions

N_2O emissions were estimated from N inputs using specific Mediterranean data for direct emissions and IPCC (2006) default procedures for indirect emissions, as described in Aguilera et al. (Submitted). Specific Mediterranean N_2O emission factors are 0.08%, 0.66% and 1.01% of applied N emitted as N_2O-N for rainfed, drip irrigation and high-water irrigation systems, respectively. N content in residues was taken from Bilandzija et al. (2012) and Rocuzzo et al. (2012). The N fixed by legumes in cover crops was included as an input using own data estimated in field (unpublished). No N fixation was assumed for soils without cover crops, whereas in extensive systems (vineyards, olives and treenuts), 20 kg N per ha were fixed by spontaneous cover crops and 45 kg by legume seeded cover crops. In intensive systems (citrus, fruits and subtropical fruits), 35 and 75 kg N were fixed in spontaneous and legume seeded cover crops, respectively. Emissions of N_2O and CH_4 from biomass burning were calculated using IPCC (2006) factors of 0.07 g N_2O and 2.7 g CH_4 emitted per kg dry matter burnt.

2.3.3 Carbon sequestration

Soil carbon sequestration is the result of a balance between carbon inputs and outputs, dependent on management practices and agro-climatic conditions, and limited in time until a new equilibrium is reached. Soil carbon balance was modeled based on the results of the meta-analysis on carbon sequestration under Mediterranean conditions by Aguilera et al. (2013a), including some unpublished data from the same study and some data from other references. Soil carbon was assumed to be in equilibrium under conventional tillage with no organic inputs. The management of tillage and organic inputs determined soil C sequestration rate in the remaining cases during the initial simulation period (0-20 years), according to the average experimental values analyzed. No tillage without cover crops was associated to a net emission of 0.13 Mg C per ha. Cover crops in woody cropping systems induced a net carbon sequestration rate of -0.27 Mg C per ha. This value can be considered conservative, for example when compared with the mean values for Spanish cover crops reported by González-Sánchez et al. (2012). It was assumed that 30.5% of the C contained in external organic inputs such as manures, composts and manufactured organic fertilizers was incorporated to the soil and thus contributed to net C accumulation. This number is the median value (N=25) of the percentage of C input contributing to net C sequestration in "organic amendments", "recommended management practices" and "slurry" categories in Aguilera et al. (2013a). We found no specific data for pruning residues, so we used the same coefficient as for the other inputs. Carbon content of residues and organic amendments was taken from Bilandzija et al. (2012). The resulting net soil carbon exchange is shown in Table 1. In the impact assessment, the model was adjusted to 100 years by considering that average C sequestration rate in the 0-100 year period was 50% of that in the initial 0-20 year period, based on the average change in published long-term modeling studies (Hansen et al. 2006; Powlson et al. 2008; Alvaro-Fuentes and Paustian 2011).

Table 1. Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) groups of fruit tree orchards crops in Spain. Data refer to one hectare and year unless otherwise stated.

	Citrus		Fruits		Subtropical		Treenuts		Vineyard		Olive	
	Conv	Org	Conv	Org	Conv	Org	Conv	Org	Conv	Org	Conv	Org
Number of interviews	5	5	12	12	5	5	6	6	7	7	7	7
Inputs												
Drip irrigation (% of cases)	60%	60%	42%	42%	80%	80%	50%	50%	0%	0%	0%	0%
Surface irrigation (% of cases)	40%	40%	33%	33%	20%	20%	0%	0%	0%	0%	0%	0%
Water use (m^3)	7280	7440	4758	4758	6500	6400	203	153	0	0	0	0
Electricity (KWh)	2300	2351	1503	1503	2054	2022	64	48	0	0	0	0
Presence of cover crops (%)	0%	64%	4%	79%	60%	93%	17%	33%	14%	29%	7%	86%
Machinery use (h)	24	18	21	28	20	31	39	44	18	18	31	28
Fuel consumption (l)	194	163	189	234	95	94	179	255	191	222	134	114
Mulching plastic (kg)	0	0	0	0	0	22	0	0	0	0	0	0
Mineral nitrogen (kg N)	284	0	91	0	216	0	42	0	8	0	67	0
Mineral phosphorus (kg P_2O_5)	127	0	63	0	64	0	27	0	25	0	24	1
Mineral potassium (kg K_2O)	137	0	118	29	431	0	39	3	33	0	33	1
Manure (Mg)	0.00	15.40	2.08	9.34	0.80	5.30	0.44	0.17	0.45	1.73	1.43	4.00
Slurry (Mg)	0.00	2.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0	15	0	20	0	0	0	0	0	21	0	21
Other organic fertilizers (kg)	0	24	3	542	0	543	0	204	0	0	0	414
Total carbon inputs (kg)	1430	3985	2264	4024	2330	2527	380	711	400	887	373	2539
Total nitrogen inputs (kg)	315	182	139	160	262	99	48	14	17	21	79	87
Synthetic pesticides (kg active matter)	23.82	0.00	12.88	0.00	7.28	0.00	1.66	0.00	3.19	0.00	1.89	0.00
Sulphur (kg)	0.00	100.00	11.13	26.99	1.20	3.15	0.00	0.00	32.14	21.86	42.86	0.00
Copper (kg)	0.00	0.00	4.63	5.92	0.00	0.00	0.00	0.00	1.45	1.28	9.71	6.00
Paraffin (kg)	3.05	6.80	7.60	1.75	0.00	0.00	1.46	1.42	0.00	0.00	0.00	0.00
Natural pesticides (kg)	0.00	5.06	0.69	7.31	0.02	4.06	0.00	2.25	0.00	0.04	0.00	0.21
Production												
Yield (Mg fresh matter, mean)	41.95	24.26	27.53	16.85	31.10	26.00	1.96	1.78	6.37	5.79	3.79	3.25
Yield (Mg fresh matter, standard deviation)	8.62	8.83	15.22	12.74	18.22	14.20	1.66	1.21	2.68	2.45	2.05	1.37
Yield (Mg dry matter. mean)	4.83	2.79	3.85	2.35	6.69	5.99	1.73	1.56	1.12	1.02	2.04	1.75
Cover crop (Mg dry matter)	0.00	1.93	0.14	2.38	2.01	2.81	0.50	1.00	0.43	0.86	0.22	2.58
Residue yield (Mg dry matter)	5.60	3.15	7.22	4.48	8.26	6.31	2.86	2.57	1.57	1.43	1.22	1.05
Residue destiny												
Open burning (Mg)	1.34	0.00	1.45	0.25	4.58	5.72	2.23	2.01	1.07	0.81	0.72	0.05
Soil incorporation (Mg)	3.03	2.46	4.19	3.24	2.20	0.00	0.00	0.00	0.15	0.30	0.23	0.76
Coproduct (Mg)	1.23	0.69	1.59	0.99	1.49	0.59	0.63	0.57	0.35	0.31	0.27	0.23
Allocation to coproduct (%)	2%	2%	4%	4%	2%	2%	10%	10%	6%	6%	5%	5%
Soil emissions												
Direct nitrous oxide (kg N_2O)	4.17	2.34	1.59	1.73	2.84	1.07	0.44	0.09	0.02	0.03	0.10	0.11
Indirect nitrous oxide (kg N_2O)	2.03	1.43	0.95	1.26	1.71	0.78	0.31	0.11	0.12	0.16	0.51	0.69
Methane (kg CH_4)	3.61	0.00	3.91	0.68	12.35	15.45	6.02	5.42	2.90	2.19	1.94	0.15
Carbon (kg C)	-276	-939	-587	-951	-396	-447	-58	-101	-34	-171	-13	-476

2.4 Sensitivity analysis

The effect of applying different methods for the estimation of C sequestration was examined in olives. The baseline scenario (*Base 100-y*) was compared with the reduction of the study time frame to 20 years (*Base 20-y*). The results were also compared with IPCC (2006) approach for 20 years and 100 years time horizons (respectively, *IPCC 100-y* and *IPCC 20-y*). In IPCC methodology C stocks are calculated for each cropping system before and after management changes on the basis of soil, climate, soil management and qualitative information on carbon input. The olive farms studied were assumed to have high-activity clay soils and warm temperate dry climate.

In many occasions wood coproducts do not have a direct economic value for the farmers, as removals are usually done by companies which are paid in wood, and tree prunings are usually consumed domestically. In order to account for this uncertainty, and to understand the physical role of fuelwood coproduction in woody systems, we quantified the effect of other allocation methods and system expansion in a sensitivity analysis of treenuts systems. These methods have different meaningfulness depending on the research question, as the functions performed by nuts and wood are of a very different nature. Treenut coproduct scenarios include product allocation (*Product*), in which all emissions are allocated to the main product; economic allocation (*Economic*); mass allocation (*Mass*); and system expansion (*Expansion*) in which wood coproducts are assumed to substitute natural gas for domestic heating purposes. The substitution of natural gas by wood was based on their calorific value. The effect of this substitution was calculated taking into account emissions during the combustion process of each type of fuel, as modelled in ecoinvent (ecoinvent Centre 2007).

3. Results and discussion

3.1 The greenhouse gas profiles of the studied systems

High variability within the studied categories, that include different study sites and management characteristics, and sometimes different crop species, leads to a high variability of yields and carbon footprints (Tables 1 and 2). Organic and conventional study cases were pair-wise selected to improve comparability, but this variability indicates that the mean values presented here should be taken with care. On the other hand, some of the sources of variation regarding the actual application of management practices and the methodological assumptions will be discussed in this section and in sections 3.2 and 3.3.

Table 2. Global warming potential of organic and conventional Spanish fruit tree orchards for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kg of product.

	Citrus		Fruits		Subtropical		Treenuts		Vineyard		Olive	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e ha^{-1})												
Machinery production	46	34	44	54	23	24	35	47	40	43	28	22
Fuel production	84	71	82	101	41	41	78	111	83	96	59	50
Fuel use	523	440	510	630	256	251	479	686	515	599	359	306
Fertilizer production	1715	71	626	149	2240	123	251	29	115	11	325	96
Direct nitrous oxide	1242	697	474	516	847	318	131	28	6	8	29	33
Indirect nitrous oxide	604	426	283	374	511	232	93	33	36	49	153	204
Pesticides	333	51	148	40	94	8	20	9	34	7	53	13
Irrigation infraestructure	201	201	141	141	278	278	156	156	0	0	0	0
Irrigation energy	1278	1306	835	835	1141	1123	36	27	0	0	0	0
Methane	90	0	98	17	309	386	150	136	72	55	49	4
Nursery	689	322	291	197	178	176	160	152	103	71	60	-24
Carbon	-507	-1722	-1075	-1744	-725	-819	-106	-185	-63	-314	-24	-873
Total	6324	1897	2597	1480	5288	2308	1531	1269	964	641	1106	-168
Total (standard deviation)	2726	346	1586	1083	4595	493	1377	860	905	1048	323	409
Product-based emissions (g CO_2e kg^{-1})												
Machinery production	1	2	2	5	1	1	31	38	6	8	10	9
Fuel production	2	3	5	9	2	2	70	90	13	17	22	20
Fuel use	14	20	31	57	13	14	428	558	83	109	132	124
Fertilizer production	38	3	24	11	122	11	106	23	18	3	85	26
Direct nitrous oxide	28	28	18	32	42	20	42	14	1	2	8	10
Indirect nitrous oxide	13	18	11	30	26	15	36	21	5	10	42	63
Pesticides	7	3	5	2	4	0	8	4	5	2	13	4
Irrigation infraestructure	5	11	7	12	14	14	78	83	0	0	0	0
Irrigation energy	30	54	28	61	62	49	13	12	0	0	0	0
Methane	2	0	6	1	11	14	69	69	10	9	12	2
Nursery	16	14	13	12	6	12	105	111	17	12	18	-4
Carbon	-12	-71	-37	-150	-6	-46	-34	-89	-4	-67	-21	-265
Total (mean)	147	83	118	94	301	113	972	955	158	106	324	-10
Total (standard deviation)	46	27	84	132	331	77	561	674	115	198	151	194
Coproduct	4	2	5	5	7	2	50	55	10	7	17	0

3.1.1 Irrigated fruits

Citrus orchards are very intensive systems with high inputs of irrigation water, fertilizers and pesticides (Table 1). They reach the highest irrigation rates and also the highest N application rates of the studied systems, with 301 and 184 kg N per ha under conventional and organic management, respectively. Consequently, they also show the highest greenhouse gases emissions rates per hectare, with 6.8 and 3.5 Mg CO_2e excluding carbon sequestration. The major decrease in organic systems occurs in fertilizer production and N_2O emissions (Table 2). When including carbon sequestration, emissions are greatly reduced in organic farms, down to 1.9 Mg CO_2e per ha, as compared to 6.3 Mg CO_2e in conventional ones, which was mainly due to high carbon inputs in the form of pruning residues and manures. Although cover crops were adopted in only 64% of the cases, internal carbon inputs represented more than half of total carbon inputs in organic citrus systems. Emissions per kg of product were relatively low in citrus fruits due to high yields, and average conventional crop emissions of 147 g CO_2e per kg were comparable with the global estimation by Nemecek et al. (2012), while emissions of organic citrus products averaged 83 g CO_2e per kg. Both values are very similar to those obtained by Pergola et al. (2013) in Sicily, despite these authors did not consider carbon sequestration. In our case, the differences with conventional management in the product-based global warming potential were only due to carbon sequestration, as the other reductions were offset by lower yields.

The analyzed “Fruit” group represents a broad range of tree species and management intensities, from heavily irrigated and fertilized cases, such as some apples and pear orchards, to rainfed fig tree orchards with almost no N input even under conventional management. This results in a great variability of greenhouse gases emissions profiles, although the general composition was similar to that of citrus production, with a relative increase in the contribution of machinery (Fig. 2a). On average, emissions per kg of product were lower under organic management but, as observed in citrus products, this only occurred when carbon sequestration was considered (Table 2).

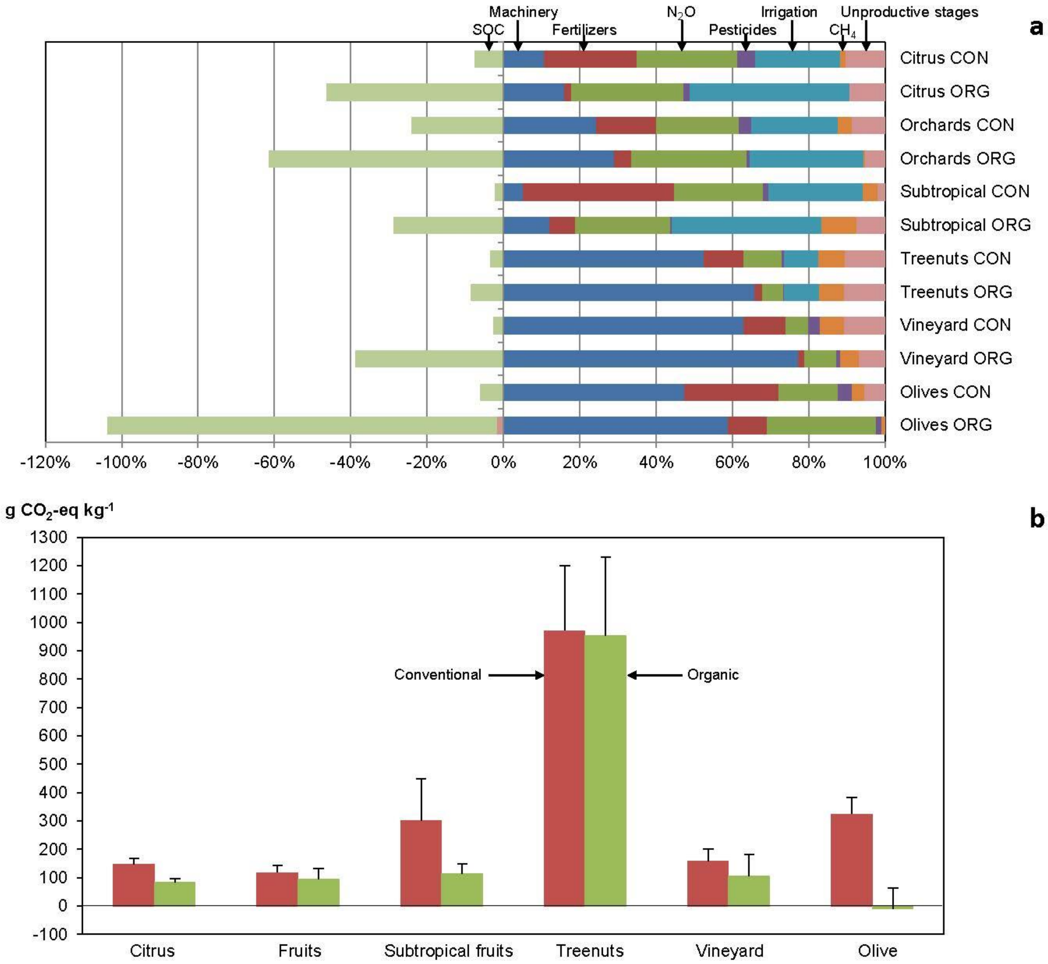

Figure 2. Global warming potential (g CO_2-eq kg^{-1}) of the six types of crop products and the two types of management (Conventional, *Con*, and Organic, *Org*) studied, expressed as the breakdown of the main processes implicated and as the net balance resulting from subtracting carbon sequestration to total emissions (means with standard errors). The components of the emission balance comprise *Unproductive stages*, which include the impact of the inputs used in the initial and final periods of the plantation; *CH_4* from biomass burning; *Pesticide* production; *N_2O*, including soil emissions, indirect emissions and biomass burning emissions, *Fertilizer production,* including fertilizer production and transport; *Machinery* production and use, including fuel production and use; and *SOC*, which accounts for the changes in soil organic carbon resulting from management practices.

Subtropical fruits repeat the same emission pattern of citrus and fruits, with a global warming potential more or less evenly distributed between fertilizer production, irrigation and nitrous oxide as the major contributors under conventional management, while irrigation, nitrous oxide and soil carbon (negative) were the most important under organic management. Yield differences between organic and conventional were not very large in subtropical fruits (Table 1), resulting in a very good performance of organic products (Fig. 2b), despite carbon sequestration did not contribute as much to the carbon footprint (Fig. 2a). It is worth noting the great variability between cases, and especially the high carbon footprint of conventional banana production, of 641 g CO_2e per kg, which was associated to a high consumption of fertilizers, and can be compared with 49 g per kg of organic bananas (data not shown) and with a global estimation of 250 g CO_2e per kg (Nemecek et al. 2012).

3.1.2 Treenuts

Treenut emissions are dominated by machinery use (Fig. 2a), as was also observed in Spanish herbaceous rainfed systems (Aguilera et al. Submitted). In fact, although our treenut sample includes 2 pairs of irrigated systems and 3 pairs of rainfed ones, the use of water and fertilizers is very low in all cases. Productivity is also low in terms of fresh matter, but similar to other rainfed crops such as olives and vineyards in terms of dry matter (Table 1). Product-based emissions of treenuts had averages of 972 and 955 g CO_2e per kg under conventional and organic management. The calculated values agree with world average emissions for almonds and hazelnuts estimated by Nemecek et al. (2012) and were lower than those associated to almond production in California reported by Venkat (2012). Higher emissions in California can be explained by a heavier use of water and pesticides and a lower productivity. In our sample, treenuts were the crop type with the worst relative performance of organic management. On average, no differences with their conventional counterparts could be observed. When compared with the other types of cropping systems, the reason seems to be a very low carbon sequestration rate in organic systems due to the low rate of adoption of soil protecting practices such as cover cropping (33%) and prunings mulching (0%). These techniques are only gradually being introduced in tree crop cultivation in Spain, as the unfavorable policy framework was preventing their expansion.

3.1.4 Vineyards

Machinery use accounted for more than 60% of the total global warming potential of vineyards (Fig. 2). The contribution of methane from open biomass burning was also important, as a result of the high amount of pruned biomass that is managed with this technique (Table 1). The main differences between conventional and organic management arise from avoided emissions from fertilizer production and increased carbon sequestration (Table 2). The absolute sequestration rate was on average very low when compared to other organic systems studied, except treenuts, due to low rates of pruning mulching and particularly of cover crop cultivation, which are the lowest of all organic groups (29%).

The average net global warming potential of 158 g CO_2e per kg of conventional grapes is in the lower range of the published values analyzed by Rugani et al. (2013), and also lower than those calculated by Villanueva-Rey et al. (2013) for conventional grape production in NW Spain. These differences are probably caused by the low input use in the studied systems. The global warming potential was reduced to 113 g CO_2e per kg under organic management, but this value is still higher than the ones reported for biodynamic farming by Villanueva-Rey et al. (op. cit.), mainly due to reduced machinery use under biodynamic management. In addition, Bosco et al. (2013) have shown that grape production emissions could be offset by carbon sequestration if adequate agronomic practices are applied, suggesting that the systems we studied were far from their mitigation potential.

3.1.5 Olives

The studied olive systems show the maximum difference in carbon footprint between conventional and organic management. The carbon footprint per kg of product averaged 324 and -10 g CO_2e, respectively (Table 2), and the composition of the profiles also differed greatly. Olive systems have a relatively high intensification degree, which implies high emissions associated to fertilizer production under conventional management and high sequestration rate under organic management (Fig. 2a). Nonetheless, they are rainfed systems with no emissions associated to irrigation and a low direct N_2O emission factor. Therefore, although absolute sequestration rates in organic citrus and fruits orchards more than doubled those of organic olive groves, the latter showed the maximum relative sequestration rates, representing an amount in terms of CO_2e similar to all the other emissions produced within the system (Fig. 2b). Remarkable soil carbon accumulation in olive groves applying recommended management practices had been previously recorded in experimental studies (e.g. Palese et al. 2013, García-Ruiz et al. 2012). Lozano-García and Parras-Alcántara (2013) showed that organic olive orchards can accumulate even more carbon in the soil than environmentally-friendly Mediterranean dehesas. To our knowledge, however, no previous study had evaluated the net effect of carbon sequestration on the total greenhouse gas budget of organic olives.

3.2 Sensitivity analysis

3.2.1 Carbon sequestration

The sensitivity analysis of carbon sequestration in olive systems (Fig. 3a) showed a very high response of the global warming potential of organic olives to the change in temporal boundaries: when these were switched from a 100-years to a 20-years time frame, the average carbon footprint of organic olives dropped from -10 to -275 g CO_2e per kg. This response was lower in the case of conventional olives, in which carbon sequestration is not an important fraction of the carbon footprint. Therefore, the average reduction in greenhouse gases emissions for organic over conventional increased from 103% to 186%, showing the great mitigation potential of organic olive farming during the first decades after organic conversion. The comparison with gross IPCC (2006) methodology yields very similar results to ours on an aggregate scale, although individual disparities exist. The effect is higher than our estimate at 20 years but lower at 100 years, as a result of the assumption of carbon content stabilization in the 20-100 years period in IPCC methodology. This assumption contrast with modeling studies that predict that carbon will continue changing, even if at a lower rate, during the whole 100-years period (Hansen et al. 2006; Powlson et al. 2008; Alvaro-Fuentes and Paustian 2011).

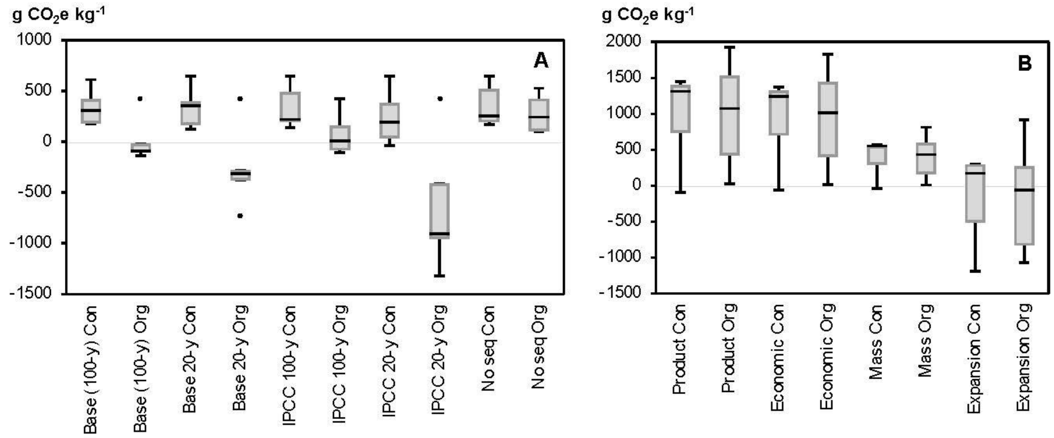

Figure 3. Sensitivity analysis of the global warming potential (g CO_2-eq kg^{-1}) of conventional (Con) and organic (Org) management of (a) olives as affected by different estimations of carbon sequestration and (b) tree nuts as affected by changes in coproduct consideration. Olive scenarios are compared to *Base (100-y)*, which represents base case sequestration, and include base case sequestration for the initial 20-year period (*Base 20-y*); IPCC sequestration estimates for 100-years and 20-years time horizons (respectively, *IPCC 100-y* and *IPCC 20-y*); and no sequestration (*No seq*). Treenut coproduct scenarios include product allocation (*Product*), in which all emissions are allocated to the main product; economic allocation (*Economic*); mass allocation (*Mass*); and system expansion (*Expansion*) in which wood coproducts are assumed to substitute natural gas for domestic heating purposes. Lines represent medians, boxes represent the 25th to 75th percentiles, non-outlier ranges (whiskers), outliers (dots) and extremes (asterisks).

3.2.2 Coproduct consideration

The sensitivity analysis of treenuts exposed in Fig. 3b reveals that the choice of the method of allocation between the main product and coproducts may have a drastic effect on the global warming potential of crop products. Allocating all emissions to nut main product (*Product*) resulted in similar emission levels as in the economic allocation between nuts and the wood coproducts (*Economic*), as the economic value of fuelwood is very low in comparison with nuts. In opposition to this lack of effect, allocation by dry matter production (*Mass*) greatly reduced the global warming potentials of both conventional and organic nut products when compared with those of main product allocation, in line with the high proportion represented by exported wood in the total biomass production of treenuts systems (Table 1). *System expansion* yielded the greatest effect on the carbon footprint, which dropped to near 0 in both types of cropping systems as a result of the consideration of fuelwood as a substitute for natural gas fuel in domestic heaters. This change in the approach of the analysis also had the largest effect on the global warming potential of cereal grains coproducing straw for forage (Aguilera et al. Submitted).

The function of energy provision through woody biomass production had seldom been recognized in the assessment of the global warming potential of fruit tree orchards, despite some studies have addressed the role of biomass accumulation in the carbon footprint (e.g. Palese et al. 2013) and other have estimated the potential for energy use of residual woody biomass (Kroodsma and Field 2006). Our results show that woody systems in Spain are already providing significant amounts of renewable energy to society, whose value in terms of saved greenhouse gases emissions may reach a similar magnitude than the emissions arising from crop cultivation.

3.3 Identifying the potential for improving the carbon footprint

Our results show a high response of the carbon footprint to differences in management, in line with other studies on greenhouse gas emissions from orchards (e.g. Mouron et al. 2006). Carbon sequestration was the greatest contributor to reduced global warming potential under organic farming in most of the studied systems (citrus, fruits, vineyards, olives), while energy-related emissions were smaller contributors. In addition, fruit tree orchards produce residues that can be burned in substitution of fossil fuels or temporally sequestered as wood products. The studied perennial systems show a large global warming mitigation potential, which has been only partially realized up to now.

3.3.1 Carbon sequestration in organic and conventional farming

The difference between organic and conventional systems in carbon sequestration rate estimated for the initial 20-years period after conversion averaged 0.29 Mg C per ha, which is below the world average of 0.45 Mg C per ha obtained by Gattinger et al. (2012) in an extensive meta-analysis, and similar to the Mediterranean average of 0.31 Mg for experimental studies reported by Aguilera et al. (2013a). In our greenhouse gases emissions balance, however, we adopted a time frame of 100 years, which resulted in an average sequestration rate of 0.14 Mg C per ha. In addition, some of our assumptions were low when compared with other published data, for example carbon sequestration associated to cover crops (González-Sánchez et al. 2012). Even with these conservative estimates, the results show how the magnitude of soil carbon sequestration in Mediterranean organic systems is often similar to that of the aggregate of all other emissions in the balance, potentially resulting in carbon neutral products. Carbon sequestration offset 9% to 102% of emissions in the studied organic categories (Fig. 2a), with an average of 38%. Negative net emission values were observed in 9 cases under organic management (6 olives, 2 vineyards and 1 fig tree) and in 2 cases under conventional management (1 carob tree and 1 vineyard). In other climates, potential carbon sequestration in orchards could be similar or even higher than in Mediterranean systems (Leinfelder et al. 2012), but its relative role in the net carbon footprint will depend on the levels of the other components of the balance.

High C sequestration rates in organic woody systems were primarily explained by the use of cover crops and the application of pruning residues to the soil. In organic farms, the combined use of both techniques resulted in an average emission of 39 g CO_2e per kg of product, compared with 129 g in those applying one of them, and 815 g in those not applying any of them. The associated changes in SOC almost always represent genuine CO_2 mitigation because the standard practices are to maintain the soil bare and burn the pruning residues. Furthermore, these practices have additional positive outcomes for erosion reduction, biodiversity enrichment and yield increase. In our study, however, many organic farmers did not apply all possible practices (65.1% of organic farms used cover crops and 57% incorporated pruning residues), suggesting a large potential for improvement.

3.3.2 Cover crops

The possibility of maintaining cover crops between trees and below tree canopies is a clear advantage for carbon sequestration in woody cropping systems. Cover cropping can increase total biomass production in the agroecosystem while reducing external resource use in mechanical or chemical weeding operations, as well as in N fertilizers. In the studied Mediterranean orchards, the carbon footprint of organic products benefit from a greater adoption degree of cover crops. The carbon footprint of the organic products cultivated with full cover cropping averaged 77 g CO_2e per kg, compared to 509 g CO_2e per kg of organic products cultivated without cover crops. Our results suggest that the increase in N_2O emissions resulting from the extra N inputs from legume cover crops is much lower than the effect on soil carbon in terms of global warming potential.

Besides the effects on mitigation, the agronomic benefits of cover cropping also extend to aspects related with adaptation and other ecological services, such as reduced soil erosion, increased water infiltration and enhanced N retention. However, there is a concern among farmers about the yield effect of water competition with the main crop, particularly in rainfed systems. Yet, these possible impacts can be overcome by an adequate timing of cover crop management operations, while the farm may have an additional output in the form of forage if the cover is grazed (Ramos et al. 2011).

3.3.3 Residual biomass

Soil application of pruning residues leads to a significant enhancement of C stocks (Palese et al. 2013) and avoids trace gas emission from biomass combustion. Our results indicate that this practice can greatly improve the greenhouse gases emissions balance of cropping systems when compared to open burning of the residues. Furthermore, additional woody residual biomass is nowadays contributing to renewable energy production in growing areas through the generalized use of thick pruning residues and of wood from plantation removals, with significant climatic benefits by preventing fossil fuel emissions (Fig. 3b). We did not specifically compare the performance of soil application with the energetic use of the residues. In any case, decision making for the destiny of the residues should differentiate their different fractions and take into account multiple factors such as the actual use, the feasibility of management operations, the nutrient content, the need to increase soil organic matter in vulnerable Mediterranean soils, and the availability of solar and wind resources as alternatives for energy production in Mediterranean areas.

On the other hand, the self-supply of nutrient and carbon inputs can be enhanced in those crops whose products are transformed by agro-industry. This is especially true for olives, as the olive oil production process generates a waste that contains virtually all nutrients of the fruit; but also applies to grapes and other fruits for the production of wine and juices, respectively. Residues can be added to the soils after adequate treatment, such as composting, resulting in a very high increase in soil carbon and soil quality

(García-Ruiz et al. 2012), while helping to avoid fertilizer use and environmental problems associated to residue management.

3.3.5 Bridging the yield gap

Lower yields reduced the climatic performance of organic systems. The type of fertilizer does not seem to represent a driver of the yield gap between organic and conventional production that we found, as other studies under Mediterranean conditions have shown that yields are not affected by fertilizer type and could even be increased by the presence of organic inputs (Palese et al. 2013; see also Aguilera et al. 2013b). Therefore, there is a need for more research in order to close this yield gap. E.g. In irrigated organic fruits this gap is mainly due to pest and disease problems, which are not well controlled. Making the most of the potential of woody system features to enhance and close nutrient cycles would also certainly help to improve yields. Efforts to enhance yields, however, should also account for the multifunctional character of woody cropping systems, which can provide not only the crop products quantified as yields, but also fuel, feed, materials and important ecological services. Adequate arrangements of available techniques to optimize this multifunctionality would greatly reduce total land requirements of organic systems, as has been studied in olive orchards (Guzmán et al. 2011). These changes require participative research and extension efforts in order to fine-tune management operations to site-specific characteristics and to spread the knowledge among farmers.

3.3.6 Reducing fossil fuel emissions in the farm and in the food chain

Fossil energy use represented a large share of emissions in the analyzed systems, despite somewhat lower than in herbaceous crops of the same area (Aguilera et al. Submitted). Mitigation measures similar to those described in Aguilera et al. (op. cit.) could also be applied in woody systems to reduce this consumption and its associated emissions, including approaches related to the reduction in energy use and those focused on the substitution of fossil energy by renewable energy. This would help to achieve significant net carbon sequestration at the farm level, which an important step since this stage usually represents a large share of emissions associated to food consumption (Rugani et al. 2013), and it is virtually the only one that can act as a carbon sink. In spite of this, further reductions in other stages of the food chain are needed in order to extend carbon neutrality to the full life cycle of food products. These actions should be taken in coherence with agricultural measures, and would include cutting fossil fuel use through energy efficiency and renewable energy in agro-industry, shortening transport distances and applying integrated waste management.

4. Conclusions

Our results draw a panoramic picture of the greenhouse gas balance of perennial woody cropping systems in Spain, as an illustrative example of a Mediterranean producing country. Distinctive emission profiles could be identified according to the types of crops, management (organic or conventional) and irrigation regimes. The degree of adoption of management techniques involving the increase of organic matter inputs to the soil exerted a profound influence on the total global warming potential through their effect on C sequestration, which can fully offset the rest of life cycle emissions of woody crop products. The results of this work show how organic management in Spain is generally more efficient in developing the climate change mitigation potential than conventional management, due to a greater degree of adoption of techniques such as cover cropping and pruning residue recycling. Nevertheless, there is a high variability in the application of these practices, and subsequently in the estimated carbon footprints of the different products. Therefore, there is still much way to further improve climatic performance through the expansion and improvement of these and other techniques. The drastic differences in C footprints observed between different management choices suggest that bold policy measures are justified. In this sense, the organic regulation in Mediterranean woody agriculture could make cover cropping mandatory and restrict the open burning of biomass. These measures should be accompanied by support for transitioning and might be extended to conventional agriculture if the huge potential for climate change mitigation by Mediterranean woody systems is to be boosted.

The present work reveals the importance of coproducts in the carbon footprint assessment of Mediterranean woody crop products, both from a methodological and from a practical point of view. The full exploitation of the multiple woody crop coproducts would mean a return to the logic of traditional multi-functional Mediterranean woody cropping systems. In the present ecological and economic crisis, there is an urgent need to guarantee the provision of multiple ecosystem services, including food, feed, energy, materials, carbon storage, biodiversity conservation and soil protection against erosion.

The new techniques and knowledge, such as pruning residue chipping tools, cover crop management expertise, composting methods, efficient biomass boilers or even more sophisticated techniques such as pyrolysis, could make the most of this logic in order to upgrade the provision of services from woody systems to the new necessities of modern society. This transition is knowledge-intensive because it has to integrate and optimize multiple functions in varied environments, so it would greatly benefit from policies supporting the production and diffusion of this information.

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Chapter 5. General discussion and conclusions

The aim of this chapter is to discuss the broader implications of the results and to identify the emerging conclusions that can be derived from the four studies of this dissertation. Therefore, the repetition of the discussion on specific results of the individual studies is herein minimized. For example, the methodological recommendations in Studies 1 and 2 will not be further discussed here.

5.1 GHG emissions in Mediterranean cropping systems

5.1.1 Soil processes

Our review and meta-analyses have shown that Mediterranean agro-climatic characteristics and management practices clearly affect soil N_2O emissions and carbon balance in these systems (Studies 1 and 2). **N_2O emissions** seem to be highly influenced by physical restrictions to microbial activity posed by typical Mediterranean climate pattern, low soil carbon content and typically low N leading to a **low N_2O emission factor (EF) in rainfed systems**. This response has also been observed in a new meta-analysis of N_2O emissions in Mediterranean cropping systems, with a higher number of studies (Cayuela et al., Under review). On the contrary, summer water availability in irrigated systems allows to increase N fertilizer rates and also increases N_2O EFs under irrigation, matching the world average EF proposed by the IPCC (2006) in high-water irrigation systems, and with intermediate EFs in drip irrigation systems. On the other hand, we observed **20% lower EFs for solid organic fertilizers** than synthetic ones, based on available N (and 63% reduction based on total N). The soil processes possibly involved in the observed differences are discussed in Study 1.

We observed a high correlation between organic inputs and **C sequestration** in Mediterranean cropping systems, with practices associated with higher **C inputs** showing consistently higher C sequestration rates. Moreover, practices relying on internal C inputs, such as cover cropping, showed a good performance in terms of C sequestration. On the other hand, **tillage** reduction or suppression only increased carbon sequestration in herbaceous systems, but decreased it in woody systems in which C inputs were also reduced (due to lower weed production). These findings are discussed in depth in Study 2.

Therefore, both **N_2O and SOC** meta-analyses indicate that the use of **organic fertilizers** substituting synthetic fertilizers could help lowering GHG emissions in Mediterranean cropping systems. There are some processes and possible interactions, however, that could be overlooked by our analyses. For example, the response of N_2O emissions and C dynamics to management practices has been shown to be highly influenced by **climate events** and the specific time in which they occur (Garland et al., 2014). As another example, it has been observed that increases in C and N stocks associated with the application of organic amendments under organic management could be accompanied by reduced **SOM stability** in Mediterranean vegetable gardens (García-Pausas et al., 2016). A possible interaction between N and C cycles affecting soil GHG production is the impact of low SOM contents on the observed low N_2O emissions in Mediterranean rainfed systems. Thus, **the long-term carbon sequestration achieved with organic inputs may compromise N_2O mitigation**. This hypothesis, however, need to be verified studying N_2O emissions as a function of organic fertilization in long-term trials, or with other experimental designs including comparisons of soils with different SOM levels. These long-term trials would also help to overcome the methodological problems related to the choice between "applied N" or "total N" for the estimation of N_2O EFs of organic fertilizers (Study 1). On the other hand, long-term **C sequestration can also be expected to promote yields** through the improvement of soil fertility (Diacono and Montemurro, 2010), which would improve yield-scaled N_2O emissions.

5.1.2 Total life cycle emissions

In our analysis of N_2O emissions and C sequestration in Mediterranean cropping systems, we identified the need for full GHG balances in order to account for the side-effects of mitigation practices on other components of the GHG balance. In the following two publications (Studies 3 and 4), we helped to fill this gap through a life cycle assessment of GHG emissions in an extensive sample of typical Mediterranean cropping systems in Spain. Our integrated LCA, employing climate-specific N_2O emission factors (that where obtained in the study 1 of this thesis) and including carbon sequestration, has shown that the GWP of Mediterranean cropping systems is dominated by **emissions related to fossil energy use** and, in the case of organic systems, also by negative emissions from **soil carbon sequestration**. These results indicate a relatively low share of N_2O emissions, in particular, and N inputs, in general, in the carbon footprint of Mediterranean crop products. This pattern is similar to that found in a tropical avocado study (Astier et al., 2014), but differs widely from studies in temperate areas, in which N_2O emissions and N inputs have a much more important role. For example, N fertilization (including fertilizer production) was related to 75% of the total C footprint of a wide range of crop products and management practice in Scotland (Hillier et al., 2009), while in our studies, the sum of N_2O (direct and indirect) and fertilizer production emissions (including P and K) represented on average 34% (Study 3) and 37% (Study 4) of the total C footprint of conventional systems (excluding C sequestration). The low relative share of fertilizer inputs in the GHG balance of the studied Mediterranean cropping systems is due to a combination of low N_2O EF and low N inputs in rainfed systems. In irrigated systems, the N_2O EF and the amount of N inputs are similar to those of temperate systems, but the relative share of fertilizers in the GHG balance is still relatively low due to the presence of other important inputs such as electricity. Fuels and electricity were also identified as the main factors contributing to the GHG balance of vineyard cropping systems in Australia (Longbottom and Petrie, 2015).

On the other hand, our study goes beyond most other crop LCAs performed under Mediterranean conditions (e.g. Romero-Gámez et al., 2014), which do not take into account carbon sequestration (e.g. Ribal et al., 2016) or employ Tier 1 IPCC N_2O EFs (Tamburini et al., 2015, Venkat, 2012) or IPCC C sequestration rates (e.g. Venkat, 2012) that have been shown a poor accuracy in Mediterranean environments (Studies 1 and 2).

5.1.3 Performance of mitigation practices

A wide range of mitigation practices has been assessed in the four studies of this dissertation. The detailed analysis of the most important processes involved in GHG emissions made possible to offer a comprehesive picture of their contribution to GHG mitigation, even though methodological constraints due to data availability hampered the evaluation of all mitigation practices in the four studies. In this section I outline the main implications of the results in terms of GHG mitigation.

Climate benefits of SOC sequestration with **reduced tillage** observed in our study (Study 2) might be offset by increased N_2O emission depending on agroclimatic conditions (Antle et al., 2012), which we could not assess in Study 1 due to lack of data. Some studies published after our meta-analysis, however, suggest that N_2O emissions in Mediterranean rainfed systems are not affected by tillage management (Tellez-Rio et al., 2015) or may have different effects depending on the type of crop (Guardia et al., 2016). In any case, low N_2O emission factors imply that, in the case of rainfed systems, the relative importance of N_2O in the GHG balance would be much lower than that of C sequestration (Guardia et al., 2016, see next section). We could not study the effect of tillage reduction in our LCAs, but other studies suggest that significant GHG benefits might be achieved mainly through reduction in fuel use (e.g. Antle et al., 2012).

Organic amendments seem as an interesting GHG mitigation practice in Mediterranean cropping systems, in particular solid organic residues which are associated to low N_2O emissions ad high C sequestration rates. As organic amendments are residues, their effectiveness for GHG mitigation would depend on their alternative fate (Sections 3.3.3 and 5.3.1).

Compost is a type of organic amendment associated to particularly low N_2O emissions, as a solid organic fertilizer (Study 1) and high C sequestration rates (Study 2). The latter effect is probably due to a greater stabilization of carbon in composted materials, which is in line with studies finding higher humification rates for compost than for other organic materials (Katterer et al., 2014), although this recalcitrance could negatively affect crop yield (Katterer et al., 2014). A recent meta-analysis has shown that the **composting** process is less emission-intensive than other organic waste management practices (Pardo et al., 2014), which would further support the use of composts to mitigate GHG.

Cover crops are another interesting mitigation practice from the point of view of soil GHG fluxes, as they significantly contribute to carbon sequestration (Study 2). As acknowledged by Poeplau and Don (2015), cover crops are particularly beneficial in terms of carbon cycling, because they represent additional carbon inputs to the soil and they are not related to yield reductions (as long as they substitute otherwise bare soils). We could not study their effect on N_2O emissions. Other studies indicate that cover crops are associated to increases in N_2O emissions, although within a context of very low N_2O emission levels (Sanz-Cobena et al., 2014a). Another benefit of cover crops is the reduction in fertilizer needs and associated production emissions, due to their N-fixation function (if they contain legumes) and catch-crop function (if they contain non-legumes) (Campiglia et al., 2011).

Rice production systems require a separate analysis because CH_4 emission in flooded soils has a very strong impact on the GHG balance (Bacenetti et al., 2016), making the balance of these systems to be very different from upland systems (Liu et al., 2014b). Our results indicate that higher organic inputs in organic systems are responsible for higher CH_4 and overall GHG emissions in these systems, a situation that is worsened by lower yields. Therefore, substantial changes are needed in Mediterranean organic rice systems in order to contribute to GHG mitigation (Bacenetti et al., 2016).

Organic farming led to higher SOC accumulation than conventional farming (Study 2), while typical organic practices such as the use of solid organic fertilizers was associated to the lowest N_2O emission rates among the studied fertilizer types (Study 1). Higher **SOC** under organic farming, which is in line with global trends (Gattinger et al., 2012), is probably linked to higher C inputs, although we could not prove this hypothesis due to the lack of data on C inputs in organic farming studies. However, acknowledged by Kallenbach et al. (2015), other mechanisms could be influencing SOM increases under organic farming: according to these authors, higher microbial growth rates and C-use efficiencies under organic farming could lead to higher C accumulation rates even with fewer C inputs and greater soil tillage. These results are in line with our finding that C sequestration rates are often superior than C inputs in the analyzed Mediterranean studies (Study 2). Other possible explanations are the enhanced root growth under low N availability (Mardanov et al., 1998) or in the presence of organic materials (Franklin et al., 2016). A characteristic emission pattern could be identified in the **LCA** of most organic cropping systems (Studies 3 and 4), with a high share of machinery-related emissions and C sequestration. The importance of fossil energy related emissions in the GHG balance of organic cropping systems was also observed in tropical avocado orchards (Astier et al., 2014) and a wide range of Mediterranean crops (Venkat, 2012). In spite of this, findings related to lower **fossil energy use** and higher energy efficiency under organic farming (Gomiero et al., 2008, 2011, Smith et al., 2015) are supported by many research studies under Mediterranean conditions, including vineyards, olives (Guzman and Alonso, 2008, Aloso and Guzman, 2010, Mohamad et al., 2014). Another characteristic of our LCA comparisons was an overall **lower GHG emissions in organic systems** than in conventional systems, particularly in the case of orchards. Other studies under Mediterranean conditions have also found relatively low emissions in organic orchards, particularly when emissions from direct land use change are taken into account (Cordes et al., 2016). On the other hand, our results indicate that **lower yields in organic farming systems** partially offset their area-based environmental benefits. Lower yield performance has also been shown the main responsible for lower overall performance in other Mediterranean organic systems (e.g. Ribal et al., 2016, Foteinis et al., 2016). Many other studies have also found lower yields under organic farming, as acknowledged by an extensive meta-analysis (Seufert et al., 2012), despite usually an overall improvement in soil quality status is also observed (Brikhofer et al., 2008).

5.1.4 Limitations of the study. The possibilities for generalization

We have studied GHG emissions in Mediterranean cropping systems through meta-analysis and LCA. **Meta-analysis** allows making generalizations from a number of individual studies, while LCA allows integrating knowledge on different processes responsible for GHG emissions to compute full C footprints of food products. Despite the potential of these approaches to synthetize and integrate scientific knowledge, however, they have **drawbacks** that have to be taken into account in order not to get misleading answers. For example, Ioannidis (2010) remarked that the true effect does not necessarily have to lie within the 95% confidence interval computed in a meta-analysis, despite that is what most meta-analysis practitioners assume. Philibert et al. (2012) studied the **quality of meta-analyses** in agronomy. They analyzed 73 meta-analyses in agronomy and found that none of them satisfied all of the eight quality criteria that they had stablished, namely: (1) explain the procedure to select papers; (2) list the references included in the analysis; (3) analysis of the variability of results; (4) analysis of the sensitivity of conclusions to changes in the dataset; (5) assessment of publication bias; (6) data weighting; (7) availability of the dataset; (8) availability of the software. Our meta-analyses (Study 1 and 2) meet most of the above criteria. We did not perform a sensitivity analysis (point 4) due to lack of space. On the other hand, despite we did not include points 3, 4, 5 and 7 in our papers, we had previously performed those analyses and made the choices on the other ones accordingly. For example, we employed random effects model and a non-parametric approach to account for the heterogeneity of our sample (point 3), while a bias-corrected method was used for computing confidence intervals (point 5). Last, the dataset was not made available in the papers (point 7), but it is included in the Appendix of this PhD dissertation.

Meta-analyses results can **vary widely depending on assumptions**, potentially calling into question the reliability of the meta-analytical findings. Hungate et al. (2009) compared four meta-analyses addressing the effect of elevated CO_2 on SOC. They found that the results differed widely because "the four meta-analyses included different studies, derived different effect size estimates from common studies, used different weighting functions and metrics of effect size, and used different approaches to address nonindependence of effect sizes", being the last factor the one with the strongest influence on differences. This variability between meta-analysis results can also be observed if we compare our results of Study 1 with those of Cayuela et al. (Under review), who also studied N_2O emission factors under Mediterranean conditions, albeit with a larger database. Despite the absolute values are different, however, the same patterns were observed regarding the influence of irrigation regime and irrigation type on N_2O emission factors, further reinforcing the soundness of our results. On the other hand, in both meta-analyses (Studies 1 and 2) we found **methodological problems in many of the primary research studies** that hindered their comparability, supporting calls for a greater effort to incorporate criteria such as large scale research, registration and sharing (of protocols, raw data, etc.), standardization, or improvement of study design into research practices (Ioannidis, 2014). These methodological problems range from uncertainty associated to spatial (Alsina et al. 2013, Ladoni et al. 2016) or temporal (Barton et al., 2008, Ju et al. 2011) measurement resolution, to the presence of control treatments, or the sampling depth. These problems are discussed in depth in the "information gap" sections of Studies 1 and 2.

Despite LCA has been subject to extensive harmonization efforts (Section 3.3.1), aimed to overcome some of the above cited drawbacks of research studies, many challenges still exist which limit the **reliability of LCA for decision support**, including representativeness of studied systems, harmonization, transparency of methodological choices, or the need for uncertainty analysis

(Brandao and Cooper, 2012). Our LCA studies (Studies 3 and 4) overcome many of the drawbacks put forward by Ioannidis (2014) and Brandao and Cooper (2012) by analyzing an extensive sample of cropping systems in Spain, as an example of a Mediterranean country, and estimating their C footprint employing harmonized LCA standards, but fine-tuning the calculation of the most sensitive processes (N_2O emissions and C sequestration) and assuring the transparency and robustness of the methods by making explicit all methodological choices and performing a sensitivity analysis. This approach allows us to contribute to the growing, but very heterogeneous, research body of food LCAs (Grunberg et al., 2010) by shedding light on general problems of agricultural LCAs, but also on the specific challenges of the assessment of GHG mitigation options in Mediterranean cropping systems. In particular, Meier et al. (2015) noticed that the comparison of organic and conventional agricultural products faced specific challenges, particularly those associated to an adequate representation of the N cycle in organic systems. Our studies overcome this limitation by a detailed modeling of N fluxes, including climate and management-specific N_2O emission factors. In addition, our results show that an adequate representation of the C cycle is also essential for a fair comparison of organic and conventional cropping systems. Last, Meier et al. (2015) also noticed the need for a consequential approach to account for different functions of the systems, which is also clearly shown in our sensitivity analyses.

5.2 The mitigation potential at the farm scale

5.2.1 Mitigation pathways in the studied systems

Our studies covered many mitigation practices currently applied in Mediterranean cropping systems. We could not assess, however, many other mitigation practices due to various methodological limitations and data availability (e.g., nitrification inhibitors, composting, rotations). In spite of this, our results shed light about the possible pathways that could be taken to mitigate GHG in the studied systems. Thus, in this section I point out possible additional measures that could be applied with potentially good results.

Overall, a **sustainable intensification** of agroecosystems appears as the most appropriate goal in most situations in order to design mitigation strategies according to our results. This is particularly true in organic systems and in low-input conventional systems. Intensive conventional systems, which are also very common in Mediterranean areas (e.g. Atilgan et al. 2008), should be more "sustainable", by combining higher use of organic materials (organic intensification) with lower use of industrial inputs (sustainable extensification, see Van Grinsven et al., 2015). On the other hand, low-impact organic systems should be more "intensified". This sustainable intensification should be based on organic fertilization, which promotes fertility while reducing GHG emissions at all levels (from soil N_2O and SOC-related emissions to fertilizer production emissions). Therefore, our results for Mediterranean cropping systems are in line with findings in other parts of the world supporting sustainable intensification as the basis for GHG mitigation in agricultural systems (Garnett et al., 2013, Bedada et al., 2016, Rusinamhodzi et al., 2016), and this strategy is further supported taking into account non-climate effects of mitigation practices (Sanz-Cobena et al., 2014b; Sanz-Cobena et al., Under Review, Section 5.2.2). Studies of energy fluxes in agroecosystem from an agro-ecological perspective also support sustainable intensification, based on the reinforcement of **low-entropy internal energy loops** (Guzman and González de Molina, 2015, Guzmán and González de Molina (Eds.), In Press).

Our results indicate that **conventional systems** should adopt several organic practices and reduce inputs consumption, given that low carbon sequestration and high emissions from inputs production were the two main processes responsible for higher GHG emissions in conventional than in organic systems (Studies 3 and 4). This "greening" of conventional cropping systems through the use of management practices such as organic fertilization, crop rotations or green manures is supported by many research studies in very diverse agroecosystems (e.g. Bhardwaj et al., 2011, Albizua et al., 2015).

On the other hand, the results also indicate that improving yield performance and reducing fossil energy use would be the most effective way to lower GHG emissions in most of the Mediterranean **organic systems** studied, along with a broader adoption of C sequestration practices in crops in which they are not widespread. The productivity in organic farms is restricted by factor such as the N supply (Berry et al. 2002), which could be improved by management practices aimed to match crop N requirements with residue N release. Some studies suggest that the productivity, multi-functionality and environmental performance of organic farming could be raised with appropriate agroecosystem designs, including diversification (Ponisio et al., 2014, Benoit et al., 2015), reconnection of crops and livestock (Garnier et al., 2016) and self-production of fuel (Aguilera, 2010, Guzmán et al., 2011). These authors also underline the potential of multifunctionality in order to meet not only mitigation goals, but also those related to adaptation and other environmental objectives (Section 5.2.2).

We have identified **farm energy use** as the major source of GHG emissions in most of the studied cropping systems. Fuel dominates in rainfed systems, while electricity production is another major source in irrigated systems. Reducing fuel consumption has benefits both for the GHG emissions and the energy balance of cropping systems. This can be achieved with practices such as reduced tillage (See Section 1.3). In the case of irrigation energy, it is necessary to take into account the source of the water in order to design energy efficiency strategies: water saving strategies based on pressurized irrigation may lead to reduced energy use when the water energy intensity is high, but to increased energy use when it is low (Jackson et al. 2011b, Sanz-Cobena et al. Under Review).

A complementary way to reduce fuel GHG emissions is to reduce fuel combustion and production emissions through the use of **alternative fuels** produced in the farm, with a special interest for organic farming systems in which fuels are the main fossil energy input (Siegmeier and Moeller, 2013). Appropriate designs are able to minimize the land cost of fuel production by enhancing eco-functional intensification (Guzmán et al., 2011, Siegmeier et al., 2015). A large potential for improvement of energy self-sufficiency and the energy return on investment (EROI) through the use of biofuels in farm machinery has been found in Mediterranean mixed systems of a low-productive area of South Spain (Aguilera, 2010). In this case, the land cost could be minimized by feeding the livestock with the high-protein biofuel production byproducts generated on-farm. These practices are especially useful for organic farms, which typically aim to minimize the use of external inputs. Indeed, most of these studies are focused on organic cropping systems. In a similar way, the carbon footprint of electricity use in agriculture could be greatly reduced by employing **renewable energy** source such as solar (Gao et al., 2013), wind, hydro or geothermal (Bayrakci and Kocar, 2012) or mixed renewable-fossil fuel systems (Carroquino et al., 2015). A more extensive application of this approach has been proposed by Bardi et al. (2013), who call for a restructuration of current agriculture, that transforms fossil fuels into food, into another one that transform renewable electricity into food.

Eliminating bare fallows in herbaceous rotations and continuous bare soils in woody systems are core practices of sustainable intensification in Mediterranean systems. Some agronomical challenges may arise from this practice, as bare soils are mainly intended to reduce water competition in these water-restricted environments. Some studies, however, indicate that bare fallows can be substituted by **legumes** in Mediterranean dryland system without, or with very low, impacts on yields of the subsequent crop (Christiansen et al., 2015). Furthermore, the expansion of legume cultivation has many additional benefits (Section 1.3.3), which could be enhanced with specific research under Mediterranean conditions. For example, the development of forage legume cultivars adapted to Mediterranean dryland conditions is a very promising field for improving the sustainability of these systems because, besides reducing the need for external fertilizers, they also have a high productivity under harsh conditions and they contribute to improve soil quality (including carbon sequestration) (Ates et al., 2014).

We could not directly study the effect of **rotations** on GHG emissions, but our data suggest that their benefits for resource use efficiency would improve the GHG balance of Mediterranean cropping systems. The results of Gonzalez-Sanchez et al. (2012), who found higher C sequestration rates in rotations when compared to monoculture in Spain (with a Mediterranean climate), also point in this direction.

Silvoarable agroforestry, the combined cultivation of herbaceous and woody crops, was quite common in traditional Mediterranean agroecosystems, but was gradually lost when they were industrialized or abandoned (Infante-Amate et al., 2016). Besides the benefits associated to multi-functionality, other advantages of silvoarable agroforestry include reduction of erosion and nitrogen leaching, and the enhancement of carbon sequestration in living biomass and landscape biodiversity (Palma et al., 2007).

The relevance of carbon sequestration and yields in our GHG emission balances suggests that **biochar**, which has been shown to improve these two variables (see Section 1.3.2), could have benefits for GHG mitigation in Mediterranean cropping systems (Vaccari et al., 2011). Results of a recent study under Mediterranean conditions (Plaza et al. 2016) suggest that biochar could promote the formation of organo-mineral complexes in organically fertilized soils, enhancing C stabilization. The combination of biochar and organic amendments, however, could also promote denitrifying activity and N_2O emissions (Sánchez-García et al., 2016). The N_2O emissions reduction effect of biochar observed at the global scale (Cayuela et al., 2014) was not found in Mediterranean cropping systems (Sánchez-García et al., 2016), although more studies are needed to verify this trend.

Other approaches aim to improve agroecosystem sustainability through **monitoring** of its properties to properly adjust management practices, such as managing livestock grazing based on grazing pressure instead of stocking rates, through the use of imaging and communication technologies to monitor grazing pressure (Sales-Baptista et al., 2016).

Other potential practices are **specific to the type of crops.** For example, emissions from **rice** production could be lowered by reducing flooding time (IPCC, 2006), employing more stabilized organic fertilizers or diversifying the rotations (Weller et al., 2016). Aerating the soil during rice cultivation period may also reduce methane emissions, but it could raise other environmental costs due to lower yields (Benecetti et al., 2016).

5.2.2 Non-GHG effects of mitigation practices

In an interconnected world reaching planetary limits, no measure addressed to solve a certain problem can be applied without considering its impact on other planetary systems and on the productive performance of the studied system itself in a context of altered environmental conditions. It is clear that the challenges posed by the agricultural trilemma (see introduction) cannot be tackled separately, but there is a need for an integrated approach to this multi-dimensional problem (Ericksen et al., 2009, Sachs et al., 2010). In particular, the implementation of agricultural GHG mitigation measures should take into account their effect on adaptation to climate change, resource consumption, and the broader socio-environmental impacts beyond the farm scale. From this perspective, the practices involving carbon sequestration seem very appropriate, as they address simultaneously mitigation, adaptation and production, having being identified as central strategies for tackling food security and climate challenges (Lal, 2004c) as well as provisioning further ecosystem services (Adhikari and Hartemink, 2016).

Thus, in the following paragraphs the main socio-environmental implications, both positive and negative, of the studied practices will be overviewed, including their direct effects for society (agronomic and economic performance, and food quality), resource use (tradeoffs with alternative uses of residual biomass, energy and water use), other components of global change (chemical pollution, biodiversity and erosion) and their interactions with adaptation to climate change.

Agronomic impacts of the studied practices vary widely, being most of them associated to potential productivity benefits, but also risks. Jacobsen et al. (2012) identified potential agronomic benefits of organic manure, early sowing enabled by minimum tillage, and an efficient weed control in Mediterranean dryland environments. **Green (and brown) manuring** in Mediterranean environments (wheat in particular) have been shown to have agronomic benefits related to weed control, delaying resistance to herbicides, reducing populations of disease organisms, and nutrition of subsequent crops (Roper et al., 2012). In a review on the potential of **organic amendments** to mitigate GHG emissions from soil, Thangarajan et al. (2013) underlined the additional co-benefits of organic amendments, but also warned about the agronomic challenges posed by their complex carbon and nitrogen dynamics. On the other hand, an agronomic implication of a mitigation practice such as **no-tillage** is the increase in **water repellency**, which is an important agronomic problem in many Mediterranean agroecosystems, such as herbaceous cropping systems in South-west Australia (Roper et al., 2015) and orchards in the Mediterranean basin (Gnzalez-Penaloza et al., 2012, Bughici and Wallace, 2016). Water repellency causes reduced water infiltration, increasing water runoff and negatively affecting yields. This process is enhanced by the accumulation of waxy compounds of

SOM in the surface of no-tilled soils, particularly in sandy soils and when there is an absence of vegetation in the soil surface. Different approaches are being studied to mitigate this problem, including synthetic (Chaichi et al., 2015) and natural (Diamantis et al., 2013) surfactants.

Ideally, the adoption of **conservation tillage under organic farming** could take advantage of the benefits of both groups of management practices observed in our results (Studies 1-4), but as observed in other parts of the world (Section 1.3.3), they arise important agronomic challenges in Mediterranean cropping systems (Sans et al., 2011, Luna et al., 2012). As we have seen, closing the yield gap appears as a major priority for **organic farming systems**. It is worth noting that sustainable practices associated to organic farming, such as rotations and organic inputs, have been consistently associated to yield increases (Branca et al., 2013), suggesting a large potential to improve yields in organic farming systems. In fact, some of the yield reductions might be because when the soil is gaining organic matter, some of the nutrients applied are being stored in the SOM instead of being available for the plants. Thus, further potential for yield increases exists in the **long term** (see Section 5.1.1). As mentioned before, it is also important to quantify the land cost of agricultural production without focusing exclusively on yields, but rather on all the **services and functions provided by agroecosystems**, including their long-term ability to support yields (sustainability). In this sense, some works have shown that appropriate management practices allow to maximize the provision of ecosystem services while minimizing the land cost (Guzmán and González de Molina, 2011, see Section 5.2.1). Ripoll-Bosch et al. (2013) accounted for the multi-functionality of agroecosystems by considering ecosystem services as an output of the system, along with yields. With the inclusion of multi-functionality, the carbon footprint of more extensive systems changed from being higher to lower than that of more intensive systems.

A good **economic performance** is essential for the feasibility of GHG mitigation practices. Sánchez et al. (2016) found that most GHG mitigation practices, including manure application, minimum tillage, rotations or crop fertilization, were financially attractive for Mediterranean farmers in NE Spain, which could help elaborating mitigation policies. Despite yields being usually lower, the economic performance of **organic systems** is generally higher than that of conventional systems, mainly due to organic price premiums (Crowder et al., 2015), even not taking into account negative externalities or ecosystem services of both systems. Moreover, despite total costs are not affected by organic farming, labor costs increase, suggesting that it can create more jobs (Crowder et al., 2015), although the opposite has been found in a Mediterranean case study (Sgroi et al., 2015). In Mediterranean areas, organic farming has also been shown to have a better economic performance, mainly through price premiums and public subsidies (Pardo et al., 2009, Sgroi et al., 2015). These better outcomes can translate into positive **socioeconomic impacts** on rural livelihoods and landscapes, particularly in less favored areas (Ronchi and Nardone, 2003), potentially contributing to reverse the abandonment trends (Testa et al., 2015). On the other hand, the dependence on price premiums and subsidies may be problematic, but they help levelling off unfair advantages of conventional systems related to the externalization of their negative impacts.

Food quality is an important food security parameter that can be affected by management. Despite the effect of single management practices on food quality have been seldom studied, the effect of organic farming has received more attention. The effects of organic farming on **food quality** usually involve higher contents of dry matter and antioxidants such as polyphenolic compounds (Baranski et al., 2014, Zalecka et al., 2014), which have been linked to a reduced risk of chronic diseases and certain cancers, and lead to longer shelf life of fresh products and better organoleptic properties (Reganold et al., 2010, Zalecka et al., 2014). Moreover, the frequency of occurrence of pesticides is greatly reduced in organic products, as well as that of the toxic metal cadmium (Baranski et al., 2014). On the other hand, the protein content of organic grains is usually reduced due to lower N availability (Casagrande et al., 2009, Campiglia et al., 2015, Mayer et al., 2015). This drawback could be tackled by the choice and specific breeding of appropriate cultivars, improved fertilization and selection of a suitable preceding crop (Reid et al., 2009, Casagrande et al., 2009, Mayer et al., 2015). The comparability of organic and conventional food products is hindered by the absence of standards in conventional production (Zalecka et al., 2014).

Competence between uses of residual biomass is a major concern for practices based on the application of organic inputs to the soil, with important implications for the GHG balance, as was discussed in Section 1.2.4 and is clearly shown by the results of the sensitivity analyses in Study 3 and Study 4. Kendall et al. (2015) also found a strong influence of allocation method and of the alternative uses of residual biomass on the carbon footprint of almond production in California. These findings indicate that a comprehensive and transparent evaluation of residual biomass destinies and their environmental implications is warranted for the design of effective mitigation strategies in Mediterranean systems.

Energy use in agriculture has been widely studied due to its central role in sustainability (Sections 1.1.1 and 1.1.2). The general trend towards lower GHG emission levels in **organic systems** observed in our LCA (Studies 3 and 4) can also be verified with respect to energy consumption, and particularly non-renewable energy use, in the same sample of farms (Alonso and Guzmán, 2010). This is not surprising, as we acknowledged the major role of emissions related to the use of fossil fuels in the GHG balance of the studied Mediterranean cropping systems. Lower non-renewable energy use has also been observed as a general global trend (Section 1.3.4) and in many other Mediterranean cropping systems (Guzmán and Alonso, 2008, Kavargiris et al., 2009, Spinelli et al., 2012). Mitigation practices related to carbon sequestration may affect energy use. As already discussed, **reduced tillage** decreases fuel consumption and associated energy use (Section 1.3). Moreover, Peltre et al. (2015) showed that **increased SOC levels** associated to repeated soil application of organic amendments was related to a reduction in draught force required for tillage. This is a very important co-benefit of soil carbon sequestration, which, this way, can contribute to reduce fossil energy use and associated GHG emissions. Another side effect of mitigation measures related to energy use are the energy requirements of **transporting** organic fertilizers, implying that a reconnection between cropland and organic matter sources (mainly manure) is required to achieve their mitigation potential while reducing energy use.

Water scarcity is a major problem in Mediterranean environments, worsened by climate change (Section 1.4.3). Drip irrigation is one of the most promising N_2O mitigation strategies (Study 1), and could lead to lower energy use when water energy intensity is high (Sanz-Cobena et al., Under Review). These savings would be related to lower water use with this type of irrigation, which thus appears as a win-win strategy in terms of the carbon-water-energy nexus. It is necessary to consider, however, that the emission and energy savings are not always achieved (Sanz-Cobena et al., Under Review), depending on water energy intensity and N input rate. On the other hand, the challenges associated to water scarcity cannot be tackled with a single technology, but requires combining biological water-saving measures with engineering solutions (Ali and Talukder, 2008) and also taking into account the potential effects of water management practices on the water cycle at the regional scale.

Chemical pollution in agriculture is mainly due to pesticide use. **Pesticide** chemical pollution has severe socio-environmental impacts, mainly related to their effects on human health and biodiversity. The degradation of chemicals in the environment also releases greenhouse gases, which are overlooked in LCA studies (Munoz et al. 2013). **No-tillage** systems usually depend on herbicides for weed control. Herbicide pollution particularly affect freshwater ecosystems (Annett et al., 2014) and some taxa, such as amphibians (Rohr et al., 2011) and corals (Jones, 2005), and can also occur in groundwater (Hallberg et al., 1989). In particular, glyphosate (along with surfactants and many other constituents of its various commercial formulae), which is the main herbicide employed in the no-tillage experiments we have reviewed, has been associated to severe damaging effects on aquatic environments (Annett et al., 2014). These harmful effects have also been observed in Mediterranean environments. For example, olive orchard soils under no tillage management with herbicides showed lower biodiversity levels than those managed with tillage (Sanchez-Moreno et al., 2015), while aquatic organisms affected by glyphosate pollution in a Mediterranean river showed many stress symptoms, despite being taxa very tolerant to pollution (Puertolas et al., 2010). Other types of chemical pollution from mitigation practices could arise from the use of **inhibitors** to reduce N_2O emissions, although the scientific information on this issue is absent. Another source of chemical pollution may come from the use of some **organic materials, such as municipal solid waste and sewage sludge**. These organic materials often contain high levels of heavy metals (Singh et al., 2008), organic contaminants (Clarke et al., 2011), or pharmaceuticals (Fent et al., 2006). The agricultural use of these residues, however, is paramount in order to close nutrient cycles and make the most of urban carbon resources in highly-populated, desert-prone Mediterranean areas, where they have shown to greatly increase soil carbon content (Study 2) and overall soil quality (Bastida et al., 2007, Roig et al., 2012). Therefore, there is a need to develop strategies to assure the safety and absence of pollutants in treated urban wastes.

Biodiversity conservation could also be an additional ecosystem service provided by **GHG mitigation** practices. Moreover, as discussed in below in this section, biodiversity management is key for adapting to climate change. In particular, organic farming systems are associated to higher biodiversity levels than conventional ones, with an overall increase in species richness averaging 30%, being greater in more intensively managed landscapes (Tuck et al., 2014). This positive effect of organic farming on biodiversity levels has been shown to be stronger in Mediterranean areas (Ponce et al., 2011), which are particularly valuable in terms of biodiversity (see Section 1.4.2). Other studies in Mediterranean environments also indicate higher biodiversity under organic farming, including intensive vineyards (Nascimbene et al., 2012) Again in this case, lower yields could compromise these benefits, according to land-sparing hypotheses (Phalan et al., 2016), although many studies suggest that appropriate agricultural management and landscape assemblages can maintain very high biodiversity levels in Mediterranean environments (Section 1.4.2).

Erosion is an important environmental problem in the Mediterranean biome, favored by the erratic rainfall pattern in these areas, which is expected to increase with climate change (Section 1.4.3). However, the effects on the net C balance are not clear (Lal 2003, Berhe et al., 2007) because the C redistribution effects of erosion have a large impact on soil C accounting (Sanderman and Chappel, 2013). Erosion reduces soil quality and negatively affect crop productivity in Mediterranean areas (De la Rosa, 2000). Erosion is particularly prevalent in woody cropping systems such as olive orchards, associated to the practice of maintaining the soil bare, with estimated erosion rates ranging from less than 1 Mg soil/ha/yr to more than 100 Mg soil/ha/yr (Gomez et al., 2011, Fleskens et al., 2007, Xiloyannis et al., 2008, Vanwalleghem et al., 2011). In turn, management practices leading to soil erosion are clearly shaped by social and institutional factors (Infante-Amate et al., 2013). The benefits of **cover crops** for soil erosion control in Mediterranean areas have been clearly demonstrated (Blavet et al., 2009, Gomez et al., 2011). Cover crops usually reduce soil loss, but not always water runoff, when compared with traditional tillage, while they reduce both impacts when compared with no-tillage without cover (Fleskens et al., 2007, Novara et al., 2011, Gómez et al., 2011). The benefits of cover crops for erosion control further reinforce the potential of this practice for climate change mitigation put forward by the results of our study (Studies 2 and 4). **No-tillage** management with bare soil (maintained with herbicides) was associated to the highest erosion rates in different experiments involving many land uses and vineyard management strategies (Blavet et al., 2009, García-Orenes et al., 2007), suggesting that soil cover, rather than tillage, is the main management factor related to soil erosion in Mediterranean woody crops. This is in line with our finding that no-tillage in woody crops is associated to lower C contents than tillage (Study 2). This protective cover can include not only cover crops but also **pruning residues** or other materials (Blavet et al., 2009).

Adaptation. The knowledge on agricultural adaptation of climate change has advanced greatly since early 1990s, when this issue began to raise researchers' attention (e.g. Rosenberg, 1992). Cumulative research evidence has shown that there is a large potential for adaptation to climate change through agricultural management practices (Iglesias et al., 1996, Rounsevell et al., 1999, Reidsma et al., 2010), and also for synergies between climate change mitigation and adaptation (Smith and Olesen, 2010, Lal et a. 2011, Campbell, 2011, Ogle et al. 2014). On the other hand, it is possible to see many adaptation measures as potential mitigation measures, particularly in terms of avoided GHG emissions due to land use change (Lobell et al., 2013). In the particular case of Mediterranean cropping systems, in which adaptation is so necessary (Jackson et al. 2011c, Section 1.4.3), the synergies between mitigation and adaptation could occur at multiple levels, and they are clearly evident in the case of organic cropping systems. Aguilera et al. (2012) outline three major ways in which organic agriculture practices contribute to adaptation in these systems: organic-based fertilization, biodiversity management and reduced external inputs. In particular, and given the current situation of widespread soil degradation (Lahmar and Ruellan, 2007), the management of soil organic matter to improve soil quality and thus long-term productivity has a central role for climate change adaptation in Mediterranean cropping systems, with multiple effects on agroecosystems leading to reduced risks (Figure 5.1).

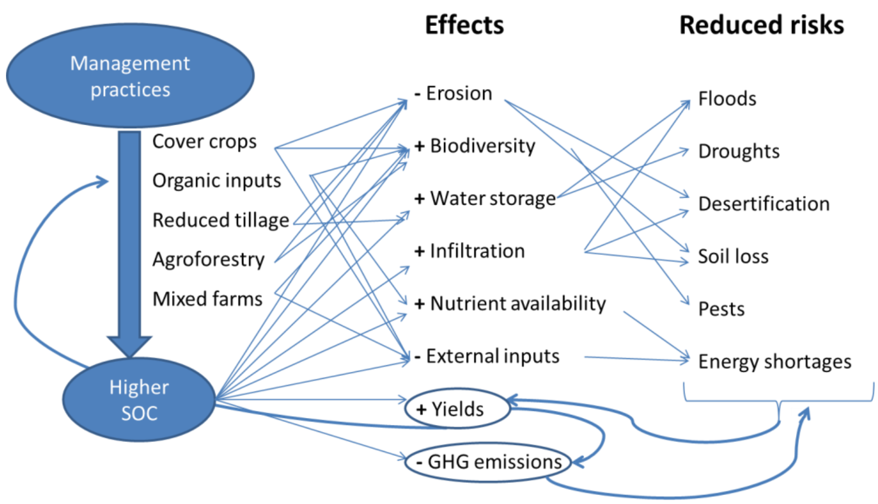

Figure 5.1. Effects of climate change mitigation practices focused on organic matter management on the response to risks associated to climate change

5.3 GHG mitigation in agriculture beyond the farm

5.3.1 Agri-food system technical measures for GHG mitigation

The impacts of food production go well beyond the agriculture phase. The industrialization of the agri-food system (AFS) has meant an enormous growth in energy demand per unit energy produced (Infante-Amate et al., 2014), and the trend continues as the processing intensity in the food chain increases, which also leads to growing health problems (Monteiro et al., 2013). Waste management (including agro-industry, urban solid waste and urban wastewater) is the central activity linking post-farm stages of the agri-food system with agricultural GHG mitigation, due to the implications for nutrients management, carbon sequestration and energy use. Moreover, waste management itself is responsible for significant GHG emissions, including biogenic CH_4 (Chiemchairisri et al., 2012), biogenic N_2O (Barton and Atwater, 2002) and GHG emissions from fossil fuel use, mainly transport (Pisoni et al., 2009). Some of the mitigation practices involve a relatively high technological component, such as the **recycling of food processing wastes** through bioenergy, biomaterial and fertilizer (Wang, 2013), or feed production (Pardo et al., 2016). Recovering the resources in organic wastes, chiefly for their ultimate use in agriculture (for animal and soils) is thus one of the major sustainability challenges of modern agri-food systems. For example, urban wastewater treatment is today a highly energy-demanding and emission-intensive activity, in which achieving carbon neutrality appears very difficult even applying best available technology (Mo and Zhang, 2012).

5.3.2 Synergies with demand-side mitigation measures

Some studies suggest that technical measures are not enough to solve the challenges faced by agriculture in the 21st century (Ericksen et al., 2009, Godfray and Garnett, 2014). Thus, mitigation measures based on **changes in demand** (or "structural" measures) are often considered as more effective, and complementary, to "supply side" measures (Foley et al., 2011, Franks and Hadingham, 2012, Garnett, 2014, Tilman et al., 2014, Bajzelj et al., 2014, Westhoek et al., 2014, Billen et al., 2015). These measures can impact agricultural emissions by shaping which, how many and where crops and animals are planted and raised. In a world facing economic degrowth (Klitgaarg and Krall, 2012), behavioral strategies to achieve lower resource consumption while maintaining or increasing human wellbeing are paramount.

Local consumption is usually claimed to reduce agri-food system GHG emissions mainly through reduced transport costs (Neira et al., 2014) and the study of the relocalization potential is gaining an increasing interest (e.g. Zumkehr and Cambell, 2015). On the other hand, some authors underline that transport emissions may be relatively low in the C footprint of food products (Weber and Matthews, 2008), and that transport reduction benefits of local food could be offset by increased emissions due to more inefficient agricultural production (Schlich and Fleissner, 2005), suggesting benefits of **global trade** for GHG mitigation. For example, off-season vegetables in cold countries have to be cultivated with heating, while the GWP would be lower importing them from warmer countries (Theurl et a. 2013). In the same study, however, emissions were reduced much further when production was **seasonal** and organic, besides local, suggesting that local consumption may have to be combined with other demand-side and farm measures to achieve its mitigation potential. Moreover, the effects of global trade go well beyond transport costs and region-specific production efficiencies: other implications involve outsourcing of GHG emissions to third countries (Peters and Hertwich, 2008, Lassaletta et al., 2014c) or problems related to global redistribution of nutrients, mainly due to feed trade. These problems are mainly associated to nutrient mining in exporting, "autotrophic" areas, and nutrient-related pollution in importing, "heterotrophic areas" (Billen et al., 2010).

Food waste is responsible for emissions related to food production and waste management. In the US, GHG emissions from food loss have been estimated at 1.4 kg CO_2-eq/cap day (Heller and Keoleian, 2015). Food waste is considered as an essential strategy for GHG mitigation in food production (Grizzetti et al., 2013, Bajzelj et al., 2014, Eberle and Fels, 2016).

The transition towards animal-based diets has enhanced the demand for land (Kastner et al., 2012), phosphorus (Metson et al., 2012) or nitrogen (Lassaletta et al., 2014b) and the levels of GHG emissions (Tilman et al., 2014). **Thus, dietary changes towards lower consumption of animal products** have been proposed as an effective mitigation strategy with additional benefits for human health and the environment, given the high carbon intensity and overall environmental costs of animal production, and the fact that the animal component in the human diet of developed region usually exceed health recommendations (González et al., 2011, Smith et al., 2013, Hallstrom et al., 2014, Tilman et al., 2014, Bajzelj et al., 2014, Heller and Keoleian 2015). In this sense, dietary shifts towards more vegetal diets are clear win-win strategies with benefits for health, resource use, the environment and also ethical. An ethical implication of high meat consumption arises from the higher land requirement of welfare-friendly production practices (Siegford et al., 2008), which means that demand has to be reduced if livestock is to be raised in an ethical way.

5.3.3 Upscaling mitigation practices

The expansion of many agricultural mitigation practices faces scalability challenges. For example, De Ponti et al. (2012) argued that the scalability of organic management was limited by nutrient availability at higher spatial scales, given the role of legumes in rotations and the availability of manure at farm and regional levels. In a similar way, scalability of organic farms in France was found to be associated to landscape reconnection of crop and livestock (Garnier et al., 2016). Thus, upscaling mitigation practices requires operating at spatial scales higher than the field or the farm, through the landscape integration of measures, which in turn would require an appropriate political framework (González de Molina, 2013).

Upscaling low-yielding management practices such as organic management implies that either cropland is expanded or the demand changes. Erb et al. (2016) simulated multiple scenarios, concluding that organic farming (at current yields) would only be able to feed the world in 2050 without cropland expansion if the global average diet was vegan or vegetarian with a high share of ruminants. Therefore, the land cost and the diet issues (Section 5.3.2) are closely interlinked and are key for the feasibility of upscaling organic farming. This also indicates that management changes alone cannot solve the problems of agriculture. In the same line, Horlings and Marsden (2011) argued that, although agro-ecological approaches could help to "feed the world", to do so they would require a radical transformation of the agri-food system, including rethinking market mechanisms, the institutional context, and science investments.

Additional functions and ecosystems services of mitigation practices (Section 5.2.2) should be taken into account when designing upscaling strategies. From this point of view, organic/agro-ecological measures seem to provide multiple socio-environmental co-benefits but, as discussed above, upscaling these practices would require coordinating efforts at multiple levels, from farms to consumers (including markets and institutions), in order to become an effective climate change mitigation option. On the other hand, the onset of the implementation of this task cannot wait until carbon sequestration, climate-specific emission factors, and a life-cycle and multi-functionality view of climate change mitigation are incorporated into global accounting protocols, and subsequent national policies are implemented. Moreover, as argued by Ostrom (2014), global-scale approaches on their own are unlikely to reduce global warming, being very vulnerable to free-rider problems. On the contrary, this author proposed a **polycentric approach** to solve this collective action problem, with an emphasis on coordinated small- to medium-scale governance units to build the strong commitment necessary to reduce individual emissions. This polycentric approach "has the main advantage of encouraging experimental efforts at multiple levels, leading to the development of methods for assessing the benefits and costs of particular strategies adopted in one type of ecosystem and compared to results obtained in other ecosystems". This is precisely the way forward that our results indicate.

5.4 General conclusions

The results of this work have revealed **distinct GHG emission patterns in Mediterranean cropping systems**. Studies 1 and 2 comprise an in-depth characterization of N_2O emissions and C sequestration, the two main processes contributing to the GHG emission balance of most cropping systems. In Studies 2 and 3 we integrated this knowledge within life cycle assessments of a wide range of Mediterranean cropping systems, which allowed us to identify the emission hotspots and to have a broader picture of management practices in the GHG emission balance.

- **Studies 1 and 2** show that different soil conditions between irrigated and rainfed crops clearly affect soil microbial processes, which control the fluxes of C (carbon dioxide, CO_2; methane, CH_4) and N (nitrous oxide, N_2O; ammonia, NH_3; nitrate, NO_3-; molecular nitrogen, N_2) in soil.
- In **Study 1** we identified distinct N_2O emissions patterns in Mediterranean cropping systems:

    -Nitrification and nitrifier denitrification are important processes for N_2O emissions in Mediterranean cropland soils, due to high aeration and low carbon. Denitrification, although also important, represents a relatively small share of N_2O emissions in many Mediterranean systems, in which water deficit is usual.

-N_2O emissions and their response to N inputs are mostly affected by **water availability**, with low emissions in rainfed systems, intermediate in drip-irrigated systems and high emissions in high-water irrigated systems. The N_2O EF of **rainfed** system is one order of magnitude lower than the average IPCC N_2O emission factor. **Drip irrigation** can be suggested as a strategy for N_2O mitigation.

-Solid **organic fertilizers** are associated to lower emissions (ca. -20%) than liquid organic and synthetic fertilizers. The choice of available N or total N applied as the denominator strongly influences the EF.

- In **Study 2** we analyzed the **response of soil organic carbon (SOC) to management practices** under Mediterranean conditions. We identified a wide spectrum of responses to the different practices.

-The **amount of soil C input** is the main driver of SOC accumulation in Mediterranean cropland soils. Management practices associated with higher C inputs are the ones with the highest C sequestration rates (organic amendments, land treatment). These practices, however, depend on external organic matter sources that may not be locally available.

-**Cover crops**, which can be widely applied in orchard systems, help maximizing C inputs based on their own biomass sources, leading to SOC increases.

-**Soil tillage reduction or elimination** is associated to significant SOC accumulation particularly in no-tillage systems. However, no-tillage with bare soil in woody systems is associated to SOC loss.

-Practices combining external amendments with cover crops or reduced tillage show a good performance in C sequestration (**combined management practices** and organic management).

-**Organic farming systems** significantly contribute to carbon sequestration, and they also show a wide variability in management practices and SOC responses. Highest SOC responses were observed in the more intensified systems.

- **Life Cycle Assessment studies 3 and 4** show that the total GHG balance is dominated by fossil energy emissions and C sequestration. Organic farming and associated practices usually reduce GHG emissions per hectare and per unit crop.

-Low N_2O EF and N inputs lead to a **low share of N_2O** in total emissions.

-**Fuel dominates** emissions in rainfed systems, while electricity is added in irrigated systems, greenhouse infrastructure in protected crops, and methane in rice.

-The magnitude of **C sequestration** is sometimes similar to that of all other emissions, leading to C-neutral systems.

- **Herbaceous crops in Spain assessed in Study 3** show very heterogeneous GHG emission patterns, depending on crop and management type

  -**Rainfed grains** (cereals and legumes) emissions are dominated by **traction energy**, while fertilizers and N_2O also represent a significant share in conventional systems.

  -**Rice** emissions are dominated by **methane**.

  -**Vegetable** emission balance is evenly distributed among irrigation, fertilizers (in conventional), N_2O and traction, with greenhouse infrastructure representing nearly 50% of emissions in protected systems.

  -GHG emissions under **organic farming** are lower than conventional both on a surface and on a product basis in all categories, with the only exception of rice.

- **Fruit tree orchards in Spain assessed in Study 4** also show heterogeneous GHG emission patterns, with an important share of carbon sequestration, particularly in organic systems.

  -**Rainfed crops** (treenuts, vineyards and olives) emissions are dominated by **traction** energy, with **fertilizers** representing a significant share in conventional systems, and **carbon** sequestration in some systems, particularly organic ones.

  -**Irrigated crops** emission balance is **evenly distributed** among irrigation, fertilizers (in conventional), N_2O and traction. Carbon sequestration offset significant emission shares, particularly in organic systems.

  -Practices involving **organic inputs to soil** (e.g. cover crops, pruning residue mulching, organic amendments) were related to lower GHG emission levels due to carbon sequestration. These practices were more widespread in organic farms, with some cases in which carbon sequestration offset all other emissions combined.

  -Despite **lower emissions** in most cases (except treenuts) the wide variability among **organic farms** indicates that many of them are far from optimizing their management, suggesting a high potential for improvement.

- These findings can be used to identify the best pathways for GHG mitigation in Mediterranean cropping systems, with important implication for policymaking and research priorities.

-Irrigation should be improved, but taking into account impacts involved along the whole life cycle of irrigation systems

-Organic inputs should be enhanced by increasing biomass production within the system (e.g. cover crops, fallow reduction) and the soil application of this biomass (e.g incorporation of pruning residues) and agro-industry biomass (e.g. olive mill waste).

-Fossil energy should be reduced through reduced energy consumption and substitution for renewables improvement.

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So many time and so many things have happened since this adventure started in 2009. During this time, I have been lucky to meet a huge number of people who have provided me all kinds of support, being my mentors in some way or another. Therefore, these paragraphs cannot possibly be exhaustive.

Scientists are said to stand on the shoulder of giants. But we can also say that they stand on the shoulders of many small people, doing small things, and contributing to change the world. This is very evident in this work, in which data comes directly from many previous research studies. The research studies employed in the meta-analysis, as well as the farms and the coefficients in the life cycle assessments, are the bricks of this thesis, and they are tied together using hundreds of other research studies cited (and not cited) in its text. Getting access to all the papers I've had to consult during this research was not always very easy given the economic barriers set by journal publishers. I've been lucky to access most of the papers through the Pablo de Olavide University service, but a large quantity of them had to be requested to the authors or retrieved from sites such as ResearchGate and SciHub. This research would have been much more difficult without those generous sharing gestures.

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The other major network in which I have participated is the Scientific network on GHG mitigation from the agroforestry sector (agriculture, livestock and forestry) (REMEDIA). It is really a privilege to have in my country such a high-level scientific network focused on the particular issue of GHG emissions in agriculture. In particular, I would like to thank Agustín del Prado, Jorge Álvaro-Fuentes, Guillermo Pardo, Raúl Moral, Fernando Estellés, David Yañez, Germán Tortosa, Alberto Sanz-Cobeña and Luis Lassaletta (again) for their effort in making it happen and for all the fruitful scientific and non-scientific exchanges we have had. Alberto, I do not have space to write all the thanks I owe you, but you know them my friend.

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The Laboratory is an improbable conjunction of experts from different research fields (history, economy, humanities, ecology, agronomy), whose combined work provides an insightful view of the history of agriculture, which challenges with hard data some of the traditional narratives, while creating tools for the development of the sustainable agriculture of the future. It's been great to work there and to meet its wonderful people, and to share all those unforgettable moments, particularly the team field trips. The invaluable and continuous support of Manuel González de Molina was a continuation of its support in SEAE. His commitment with agroecology and his belief in me made this thesis possible. David Soto, Roberto García, Antonio Herrera, Antonio Cid, Juan Infante, Inma

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Aunque el origen último siempre está mucho más atrás, por razones operativas podríamos decir que el precursor de esta tesis fue el máster de Agroecología que cursé en Baeza, donde me empapé de la experiencia de tantxs profesorxs y alumnxs, y de la propia experiencia que suponía el encierro en ese claustro medieval junto a gente tan diversa. Ahí encajaron de repente dentro de un marco teórico robusto muchas de las ideas que vagaban por mi cabeza desordenadas: el puzzle resuelto mostraba un paisaje terriblemente crudo, donde la injusticia en el campo no sólo existía entre clases sino también entre géneros, territorios, generaciones, cosmovisiones y especies. Pero también mostraba una caja de herramientas esperanzadora, que insinuaba maneras en las que la gente podía trabajar junto con la gente y con la naturaleza, en lugar de contra ellas. También ahí nacieron muchas amistades que hoy me siguen enriqueciendo. Irene, Paco, Isa, Alba, Patri, Marcus, y todas las demás. Os siento muy cerca aunque estéis lejos. Qué lujo conoceros y viviros.

Lxs agricultorxs han generado la información de gran parte de esta tesis. La actividad agraria del mundo real que se tiene que enfrentar a sus imposiciones y actuar en consecuencia diariamente, digan lo que digan los modelos. Muchas gracias por haber compartido la información, y por hacer lo que hacéis.

Las jornadas doctorales del programa de doctorado de Estudios Medioambientales de la UPO, en la propia UPO y en Panamá, me han permitido compartir experiencias de la tesis con otrxs doctorandxs, lo cual aporta mucho porque el feedback no es sólo académico sino también de experiencia vital. De igual modo, agradezco también a las personas y colectivos que me han invitado a participar en sus espacios divulgando los resultados de la tesis más allá del ámbito científico: Juanlu del KHA, José Luis de CERAI, Itziar de la US, Gustavo de Soberanía Alimentaria, Esther de La Nave, Inma de la UPO, Alberto de la UPMC, Esther de La ReVerde, Rocío de Latidos, Julia de EcoSostenibleWine, Elena de la UCO. En todos estos lugares, y en tantos otros cursos, jornadas, congresos y demás eventos, conocí gente estupenda con la que pasé grandes ratos. Y da mucha alegría cuando volvemos a juntarnos.

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Appendix A1. Supplementary data of Study 1

Table A1.1 Basic data of analyzed datasets, previous to aggregation. Fertilizer abbreviations: U: Urea; AN: Ammonium Nitrate; AS: Ammonim Sulphate; AP: Ammonium phosphate; NPK: Compound fertilizers; LCC: Legume cover crops; CC: Non-legum cover crops; CR: Crop residues; CR* Crop residues not accounted for; DPS: Digested Pig Slurry; PS: Pig Slurry; SM: Sheep Manure; PCM: Pasteurized chicken manure; None: No fertilizers applied

Dataset number	Reference	Crop	Crop type	Fertilizer	Fertilizer type	Irrigation	Irrigation type
1	Barton et al. (2011)	Lupin	Legume	LCC	Organic (solid)	Rainfed	Rainfed
2	Barton et al. (2011)	None	None	None	None	Rainfed	Rainfed
3	Barton et al. (2010)	Canola	Other	U	Synthetic	Rainfed	Rainfed
4	Barton et al. (2010)	Canola	Other	None	None	Rainfed	Rainfed
5	Barton et al. (2008)	Wheat	Winter cereals	U	Synthetic	Rainfed	Rainfed
6	Barton et al. (2008)	Wheat	Winter cereals	None	None	Rainfed	Rainfed
7	Garland et al. (2011)	Legume mixture	Legume	LCC	Organic (solid)	Rainfed	Rainfed
8	Garland et al. (2011)	Legume mixture	Legume	LCC	Organic (solid)	Rainfed	Rainfed
9	Garland et al. (2011)	Vineyard	Vineyard	LCC + U-AN	Organic + Synthetic	Drip	Low-Water
10	Garland et al. (2011)	Vineyard	Vineyard	LCC + U-AN	Organic + Synthetic	Drip	Low-Water
11	Kallenbach et al. (2010)	Tomato	Horticulture	LCC + NPK + AS	Organic + Synthetic	Furrow	High-Water
12	Kallenbach et al. (2010)	Tomato	Horticulture	LCC + NPK	Organic + Synthetic	Drip	Low-Water
13	Kallenbach et al. (2010)	Tomato	Horticulture	NPK + AS	Synthetic	Furrow	High-Water
14	Kallenbach et al. (2010)	Tomato	Horticulture	NPK + AS	Synthetic	Drip	Low-Water
15	Kong et al. (2009)	Maize	Maize	Compost + LCC + CR*	Organic (solid)	Furrow	High-Water
16	Kong et al. (2009)	Maize	Maize	U-AS + LCC + CR*	Organic + Synthetic	Furrow	High-Water
17	Kong et al. (2009)	Maize	Maize	U-AS + CR*	Synthetic	Furrow	High-Water
18	Kong et al. (2009)	Maize	Maize	Compost + LCC +	Organic (solid)	Furrow	High-Water

				CR*			
19	Kong et al. (2009)	Maize	Maize	U-AS + LCC + CR*	Organic + Synthetic	Furrow	High-Water
20	Kong et al. (2009)	Maize	Maize	U-AS + CR*	Synthetic	Furrow	High-Water
21	Lee et al. (2009)	Maize	Maize	U-AN + NPK + CR*	Synthetic	Furrow	High-Water
22	Lee et al. (2009)	Maize	Maize	U-AN + NPK + CR*	Synthetic	Furrow	High-Water
23	Lee et al. (2009)	Sunflower	Other	U-AN + CR*	Synthetic	Furrow	High-Water
24	Lee et al. (2009)	Sunflower	Other	U-AN + CR*	Synthetic	Furrow	High-Water
25	Lee et al. (2009)	Chickpea	Legume	CR*	None	Rainfed	Rainfed
26	Lee et al. (2009)	Chickpea	Legume	CR*	None	Rainfed	Rainfed
27	López-Fernández et al. (2007)	Maize	Maize	MSW Compost	Organic (solid)	Sprinkler	High-Water
28	López-Fernández et al. (2007)	Maize	Maize	SM Compost	Organic (solid)	Sprinkler	High-Water
29	López-Fernández et al. (2007)	Maize	Maize	PS surface	Organic (liquid)	Sprinkler	High-Water
30	López-Fernández et al. (2007)	Maize	Maize	PS inyected	Organic (liquid)	Sprinkler	High-Water
31	López-Fernández et al. (2007)	Maize	Maize	U	Synthetic	Sprinkler	High-Water
32	López-Fernández et al. (2007)	Maize	Maize	None	None	Sprinkler	High-Water
33	López-Fernández et al. (2007)	Bare	None	MSW Compost	Organic (solid)	Sprinkler	High-Water
34	López-Fernández et al. (2007)	Bare	None	SM Compost	Organic (solid)	Sprinkler	High-Water
35	López-Fernández et al. (2007)	Bare	None	PS surface	Organic (liquid)	Sprinkler	High-Water
36	López-Fernández et al. (2007)	Bare	None	PS inyected	Organic (liquid)	Sprinkler	High-Water
37	López-Fernández et al. (2007)	Bare	None	U	Synthetic	Sprinkler	High-Water
38	López-Fernández et al. (2007)	Bare	None	None	None	Sprinkler	High-Water
39	Heller et al. (2010)	Maize	Maize	NPK + CR	Organic + Synthetic	Drip	Low-Water
40	Heller et al. (2010)	Maize	Maize	NPK	Synthetic	Drip	Low-Water
41	Meijide et al. (2007)	Maize	Maize	PS	Organic (liquid)	Sprinkler	High-Water
42	Meijide et al. (2007)	Maize	Maize	DPS	Organic (liquid)	Sprinkler	High-Water
43	Meijide et al. (2007)	Maize	Maize	U	Synthetic	Sprinkler	High-Water
44	Meijide et al. (2007)	Maize	Maize	DPS Compost + U	Organic + Synthetic	Sprinkler	High-Water
45	Meijide et al. (2007)	Maize	Maize	MSW Compost + U	Organic + Synthetic	Sprinkler	High-Water
46	Meijide et al. (2007)	Maize	Maize	None	None	Sprinkler	High-Water

47	Meijide et al. (2009)	Barley	Winter cereals	CR Compost + SS	Organic (solid)	Rainfed	Rainfed
48	Meijide et al. (2009)	Barley	Winter cereals	MSW	Organic (solid)	Rainfed	Rainfed
49	Meijide et al. (2009)	Barley	Winter cereals	PS	Organic (liquid)	Rainfed	Rainfed
50	Meijide et al. (2009)	Barley	Winter cereals	DPS	Organic (liquid)	Rainfed	Rainfed
51	Meijide et al. (2009)	Barley	Winter cereals	U	Synthetic	Rainfed	Rainfed
52	Meijide et al. (2009)	Barley	Winter cereals	None	None	Rainfed	Rainfed
53	Petersen et al. (2006)	Rotation	Other	M + LCC + D	Organic (solid)	High-water	High-Water
54	Petersen et al. (2006)	Rotation	Other	NPK + M + LCC + D	Organic + Synthetic	High-water	High-Water
55	Sánchez-Martín et al. (2008b)	Melon	Horticulture	AS	Synthetic	Drip	Low-Water
56	Sánchez-Martín et al. (2008b)	Melon	Horticulture	None	None	Drip	Low-Water
57	Sánchez-Martín et al. (2008b)	Melon	Horticulture	AS	Synthetic	Furrow	High-Water
58	Sánchez-Martín et al. (2008b)	Melon	Horticulture	None	None	Furrow	High-Water
59	Sánchez-Martín et al. (2010a)	Melon	Horticulture	DPS	Organic (liquid)	Furrow	High-Water
60	Sánchez-Martín et al. (2010a)	Melon	Horticulture	None	None	Furrow	High-Water
61	Sánchez-Martín et al. (2010a)	Melon	Horticulture	DPS	Organic (liquid)	Drip	Low-Water
62	Sánchez-Martín et al. (2010a)	Melon	Horticulture	None	None	Drip	Low-Water
63	Sánchez-Martín et al. (2010b)	Onion	Horticulture	OM	Organic (solid)	Micro-sprinkler	High-Water
64	Sánchez-Martín et al. (2010b)	Onion	Horticulture	DPS	Organic (liquid)	Micro-sprinkler	High-Water
65	Sánchez-Martín et al. (2010b)	Onion	Horticulture	U	Synthetic	Micro-sprinkler	High-Water
66	Sánchez-Martín et al. (2010b)	Onion	Horticulture	None	None	Micro-sprinkler	High-Water
67	Sánchez-Martín et al. (2010b)	Onion	Horticulture	OM	Organic (solid)	Micro-sprinkler	High-Water
68	Sánchez-Martín et al. (2010b)	Onion	Horticulture	DPS	Organic (liquid)	Micro-sprinkler	High-Water
69	Sánchez-Martín et al. (2010b)	Onion	Horticulture	U	Synthetic	Micro-sprinkler	High-Water
70	Sánchez-Martín et al. (2010b)	Onion	Horticulture	None	None	Micro-sprinkler	High-Water
71	Steenwerth et al. (2008)	Vineyard cover	Vineyard	CC	Organic (solid)	Rainfed	Rainfed
72	Steenwerth et al. (2008)	Vineyard cover	Vineyard	CC	Organic (solid)	Rainfed	Rainfed
73	Steenwerth et al. (2008)	Vineyard cover	Vineyard	None	None	Rainfed	Rainfed
74	Townsend-Small et al. (2011)	Maize	Maize	AP	Synthetic	Furrow	High-Water
75	Townsend-Small et al. (2011)	Rotation	Horticulture	AS	Synthetic	Drip	Low-Water
76	Vallejo et al. (2005)	Maize	Maize	PS surface	Organic (liquid)	Sprinkler	High-Water
77	Vallejo et al. (2005)	Maize	Maize	PS inyected	Organic (liquid)	Sprinkler	High-Water
78	Vallejo et al. (2005)	Maize	Maize	None	None	Sprinkler	High-Water
79	Vallejo et al. (2006)	Potato	Other	PS inyected	Organic (liquid)	Sprinkler	High-Water

80	Vallejo et al. (2006)	Potato	Other	DPS	Organic (liquid)	Sprinkler	High-Water
81	Vallejo et al. (2006)	Potato	Other	DPS Compost	Organic (solid)	Sprinkler	High-Water
82	Vallejo et al. (2006)	Potato	Other	U + MSW	Organic + Synthetic	Sprinkler	High-Water
83	Vallejo et al. (2006)	Potato	Other	U	Synthetic	Sprinkler	High-Water
84	Vallejo et al. (2006)	Potato	Other	None	None	Sprinkler	High-Water

Table A1.1 (Continued) Basic data of analyzed datasets, previous to aggregation

Dataset number	SOC	Average temperature	Water Average precipitation	Irrigation	N inputs (kg N/ha) Total N	Available N	N_2O emissions (kg N_2O-N/ha) Control	Treatment	Year-basis	Emission Factor (%) EF	EF-year basis
1	9.8	18.3	355	0	67		0.13	0.13	0.13		
2	9.8	18.3	355	0	0	0		0.13	0.14		
3	9.8	18.3	267	0	75	75	0.08	0.13	0.13	0.06%	0.06%
4	9.8	18.3	267	0	0	0		0.08	0.08		
5	9.8	18.3	358	0	100	100	0.09	0.11	0.11	0.02%	0.02%
6	9.8	18.3	358	0	0	0		0.09	0.09		
7		20.9	427	160	47			0.07	0.13		
8		20.9	427	160	47			0.11	0.21		
9		20.9	427	160	52	52		0.09	0.17		
10		20.9	427	160	52	52		0.14	0.26		
11	13.0	15.8	480	886	220	220		5.83	5.83		
12	13.0	15.8	480	381	220	220		3.52	3.52		
13	11.0	15.8	480	886	120	120		2.75	2.75		
14	11.0	15.8	480	381	120	120		1.50	1.50		
15	10.0	17.0	462	850	473			0.75	1.93		
16	7.6	17.0	462	740	190			0.56	1.44		
17	8.1	17.0	462	670	280	280		2.21	5.68		
18	10.3	17.0	462	850	473			0.18	0.46		
19	7.9	17.0	462	740	190			0.38	0.98		
20	8.1	17.0	462	670	280	280		1.13	2.90		
21	9.7	16.1	564	338	244	244		8.53	8.53		

22	8.9	16.1	564	338	244	244		3.83	3.83		
23	9.7	16.1	564	124	90	90		3.84	3.84		
24	8.9	16.1	564	124	90	90		2.91	2.91		
25	9.7	16.1	564	0	0	0		2.03	4.07		
26	8.9	16.1	564	0	0	0		1.28	2.57		
27	8.2	13.5	460	488	425	170	2.91	3.77	6.88	0.50%	0.91%
28	8.2	13.5	460	488	425	170	2.91	3.71	6.77	0.46%	0.84%
29	8.2	13.5	460	488	170	170	2.91	4.66	8.50	1.02%	1.86%
30	8.2	13.5	460	488	170	170	2.91	5.08	9.27	1.27%	2.32%
31	8.2	13.5	460	488	170	170	2.91	5.89	10.75	1.80%	3.29%
32	8.2	13.5	460	488	0	0		2.91	5.31		
33	8.2	13.5	460	488	425	170	2.99	3.12	5.69	0.08%	0.15%
34	8.2	13.5	460	488	425	170	2.99	4.54	8.29	0.94%	1.72%
35	8.2	13.5	460	488	170	170	2.99	3.47	6.33	0.29%	0.53%
36	8.2	13.5	460	488	170	170	2.99	4.33	7.90	0.76%	1.39%
37	8.2	13.5	460	488	170	170	2.99	3.71	6.77	0.42%	0.77%
38	8.2	13.5	460	488	0	0		2.99	5.46		
39	10.3	20.0	431		302	302		0.94	0.94		
40	10.3	20.0	431		240	240		0.77	0.77		
41	8.9	13.5	460	485	175	175	5.98	8.27	20.82	1.31%	3.29%
42	8.9	13.5	460	485	175	175	5.98	7.70	19.38	0.98%	2.47%
43	8.9	13.5	460	485	175	175	5.98	8.57	21.57	1.48%	3.73%
44	8.9	13.5	460	485		175	5.98	9.28	23.36	1.89%	4.75%
45	8.9	13.5	460	485		175	5.98	7.13	17.95	0.66%	1.65%
46	8.9	13.5	460	485	0	0		5.98	15.05		
47	8.2	13.2	430	0	313	125	0.20	0.37	0.41	0.14%	0.15%
48	8.2	13.2	430	0	313	125	0.20	0.27	0.29	0.05%	0.05%
49	8.2	13.2	430	0	125	125	0.20	0.32	0.34	0.09%	0.10%
50	8.2	13.2	430	0	125	125	0.20	0.35	0.38	0.12%	0.13%
51	8.2	13.2	430	0	125	125	0.20	0.35	0.38	0.11%	0.12%
52	8.2	13.2	430	0	0	0		0.20	0.22		
53					117			2.73	2.73		
54					286			4.94	4.94		

55	8.2	13.2	430	194	175	175	0.90	1.65	4.30	0.43%	1.12%
56	8.2	13.2	430	194	0	0		0.90	2.35		
57	8.2	13.2	430	348	175	175	2.95	5.23	13.64	1.30%	3.40%
58	8.2	13.2	430	348	0	0		2.95	7.69		
59	8.2	13.2	430	325	175	175	2.80	3.14	7.64	0.19%	0.47%
60	8.2	13.2	430	325	0	0		2.80	6.81		
61	8.2	13.2	430	205	175	175	0.70	2.27	5.52	0.90%	2.18%
62	8.2	13.2	430	205	0	0		0.70	1.70		
63	8.2	13.2	608	607	275	110	0.16	0.94	0.94	0.71%	0.71%
64	8.2	13.2	608	607	110	110	0.16	1.05	1.05	0.81%	0.81%
65	8.2	13.2	608	607	110	110	0.16	1.00	1.00	0.76%	0.76%
66	8.2	13.2	608	607	0	0		0.16	0.16		
67	8.2	13.2	50	607	275	110	0.35	0.83	0.83	0.44%	0.44%
68	8.2	13.2	50	607	110	110	0.35	1.01	1.01	0.60%	0.60%
69	8.2	13.2	50	607	110	110	0.35	0.93	0.93	0.53%	0.53%
70	8.2	13.2	50	607	0	0		0.35	0.35		
71	11.0	13.5	460	0			0.47	0.57	0.57		
72	9.5	13.5	460	0			0.47	0.70	0.70		
73	7.2	13.5	460	0	0	0		0.47	0.47		
74					160	160		2.20	2.20		
75					70	70		1.20	1.20		
76	8.9	13.5	460	403	200	200	4.60	7.80	13.24	1.60%	2.72%
77	8.9	13.5	460	403	200	200	4.60	10.50	17.83	2.95%	5.01%
78	8.9	13.5	460	403	0	0		4.60	7.81		
79	8.2	13.2	430	460	175	175	3.69	5.62	13.68	1.10%	2.68%
80	8.2	13.2	430	460	175	175	3.69	4.69	11.41	0.57%	1.39%
81	8.2	13.2	430	460	300	175	3.69	6.41	15.60	1.55%	3.77%
82	8.2	13.2	430	460	300	175	3.69	5.65	13.75	1.12%	2.73%
83	8.2	13.2	430	460	175	175	3.69	7.31	17.79	2.07%	5.04%
84	8.2	13.2	430	460	0	0		3.69	8.98		

Table A1.2. Cumulative N_2O emissions of control, organic, and synthetic fertilizer treatments of the aggregated datasets employed in the meta-analysis. The dataset codes refer to Table A1.1

	Control		Organic fertilizer				Synthetic fertilizer		
Reference	kg N_2O-N/ha	SD	Dataset code	Category	kg N_2O-N/ha	SD	Dataset code	kg N2O-N/ha	SD
López-Fernández et al. (2007)	2.91	0.69	29, 30	Liquid	3.74	0.85	31	5.89	0.54
López-Fernández et al. (2007)	2.99	0.38	35, 36	Liquid	3.83	0.33	37	3.71	0.36
Meijide et al. (2007)	5.98	0.69	41	Liquid	7.99	1.19	43	8.57	0.95
Meijide et al. (2009)	0.20	0.05	49	Liquid	0.33	0.08	51	0.35	0.11
Sánchez-Martín et al. (2010b)	0.16	0.14	64	Liquid	1.05	0.10	65	1.00	0.18
Sánchez-Martín et al. (2010b)	0.35	0.17	68	Liquid	1.01	0.12	69	0.93	0.15
Vallejo et al. (2006)	3.69	0.36	79, 80	Liquid	5.16	0.42	83	7.31	0.39
López-Fernández et al. (2007)	2.91	0.69	27, 28	Solid	3.74	0.85	31	5.89	0.54
López-Fernández et al. (2007)	2.99	0.38	33, 34	Solid	3.83	0.33	37	3.71	0.36
Meijide et al. (2009)	0.20	0.05	47, 48, 50	Solid	0.32	0.06	51	0.35	0.11
Sánchez-Martín et al. (2010)	0.16	0.14	63	Solid	0.94	0.04	65	1.00	0.18
Sánchez-Martín et al. (2010)	0.35	0.17	67	Solid	0.83	0.03	69	0.93	0.15
Vallejo et al. (2006)	3.69	0.36	81	Solid	6.41	0.57	83	7.31	0.39
López-Fernández et al. (2007)	2.91	0.69	27, 28, 29, 30	All organic	4.31	0.81	31	5.89	0.54
López-Fernández et al. (2007)	2.99	0.38	33, 34, 35, 36	All organic	3.87	0.65	37	3.71	0.36
Meijide et al. (2007)	5.98	0.69	41	All organic	7.99	1.19	43	8.57	0.95
Meijide et al. (2009)	0.20	0.05	47, 48, 49, 50	All organic	0.33	0.07	51	0.35	0.11
Sánchez-Martín et al. (2010)	0.35	0.17	63, 64	All organic	0.92	0.08	31	0.93	0.15
Sánchez-Martín et al. (2010)	0.16	0.14	67, 68	All organic	1.00	0.07	37	1.00	0.18
Vallejo et al. (2006)	3.69	0.36	79, 80, 81	All organic	5.57	0.47	83	7.31	0.39

Table A1.3. N_2O emission factors of control, organic, and synthetic fertilizer treatments of the aggregated datasets employed in the meta-analysis. The dataset codes refer to Table A1.1

	Organic fertilizer						Synthetic fertilizer			
	Dataset code	EF	SD	EF Total N	SD	n	Dataset code	EF	SD	n

López-Fernández et al. (2007)	29, 30	0.49%	0.50%	0.20%	0.20%	6	31	1.80%	0.36%	3
López-Fernández et al. (2007)	35, 36	0.49%	0.21%	0.20%	0.08%	6	37	0.42%	0.22%	3
Meijide et al. (2007)	41	1.15%	0.56%	1.15%	0.56%	3	43	1.48%	0.47%	3
Meijide et al. (2009)	49	0.11%	0.06%	0.11%	0.06%	3	51	0.11%	0.07%	3
Sánchez-Martín et al. (2010b)	64	0.81%	0.11%	0.81%	0.11%	3	65	0.76%	0.15%	3
Sánchez-Martín et al. (2010b)	68	0.60%	0.13%	0.60%	0.13%	3	69	0.53%	0.15%	3
Vallejo et al. (2006)	79, 80	0.84%	0.23%	0.84%	0.23%	3	83	2.07%	0.21%	3
López-Fernández et al. (2007)	27, 28	0.49%	0.50%	0.20%	0.20%	6	31	1.80%	0.36%	3
López-Fernández et al. (2007)	33, 34	0.49%	0.21%	0.20%	0.08%	6	37	0.42%	0.22%	3
Meijide et al. (2009)	47, 48, 50	0.09%	0.05%	0.04%	0.02%	6	51	0.11%	0.07%	3
Sánchez-Martín et al. (2010)	63	0.71%	0.09%	0.28%	0.04%	3	65	0.76%	0.15%	3
Sánchez-Martín et al. (2010)	67	0.44%	0.11%	0.17%	0.04%	3	69	0.53%	0.15%	3
Vallejo et al. (2006)	81	1.55%	0.27%	0.91%	0.16%	3	83	2.07%	0.21%	3
López-Fernández et al. (2007)	27, 28, 29, 30	0.82%	0.47%	0.67%	0.32%	12	31	1.80%	0.36%	3
López-Fernández et al. (2007)	33, 34, 35, 36	0.51%	0.32%	0.37%	0.26%	12	37	0.42%	0.22%	3
Meijide et al. (2007)	41	1.15%	0.56%	1.15%	0.56%	6	43	1.48%	0.47%	3
Meijide et al. (2009)	47, 48, 49, 50	0.10%	0.05%	0.07%	0.04%	12	51	0.11%	0.07%	3
Sánchez-Martín et al. (2010)	63, 64	0.52%	0.12%	0.39%	0.09%	6	31	0.53%	0.15%	3
Sánchez-Martín et al. (2010)	67, 68	0.76%	0.10%	0.55%	0.07%	6	37	0.76%	0.15%	3
Vallejo et al. (2006)	79, 80, 81	1.08%	0.24%	0.86%	0.20%	9	83	2.07%	0.21%	3

Table A1.4. Response ratios of N_2O cumulative emissions and emissions factors of comparisns between organic and synthetic fertilizers. Aggregated datasets employed in the meta-analysis. The dataset codes refer to Table A1.1

	Dataset code		Cumulative emissions			Emission Factor		
Reference	Organic	Synthetic	RR	Ln RR	V (Ln RR)	RR	Ln RR	V (Ln RR)
López-Fernández et al. (2007)	29, 30	31	0.83	-0.19	0.01	0.64	-0.45	0.05
López-Fernández et al. (2007)	35, 36	37	1.05	0.05	0.02	1.27	0.24	0.36
Meijide et al. (2007)	41	43	0.93	-0.07	0.01	0.77	-0.26	0.09
Meijide et al. (2009)	49	51	0.97	-0.03	0.05	0.96	-0.04	0.19
Sánchez-Martín et al. (2010b)	64	65	1.05	0.05	0.01	1.06	0.06	0.02
Sánchez-Martín et al. (2010b)	68	69	1.09	0.08	0.01	1.14	0.13	0.04

Vallejo et al. (2006)	79, 80	83	0.71	-0.35	0.00	0.40	-0.91	0.03
López-Fernández et al. (2007)	27, 28	31	0.63	-0.45	0.02	0.27	-1.30	0.28
López-Fernández et al. (2007)	33, 34	37	1.03	0.03	0.00	1.18	0.16	0.72
Meijide et al. (2009)	47, 48, 50	51	0.83	-0.18	0.01	0.85	-0.16	0.16
Sánchez-Martín et al. (2010)	63	65	0.94	-0.06	0.01	0.93	-0.07	0.02
Sánchez-Martín et al. (2010)	67	69	0.89	-0.11	0.01	0.83	-0.19	0.05
Vallejo et al. (2006)	81	83	0.88	-0.13	0.00	0.75	-0.29	0.01
López-Fernández et al. (2007)	27, 28, 29, 30	31	0.73	-0.31	0.01	0.46	-0.79	0.12
López-Fernández et al. (2007)	33, 34, 35, 36	37	1.04	0.04	0.01	1.23	0.20	0.43
Meijide et al. (2007)	41	43	0.93	-0.07	0.01	0.77	-0.26	0.09
Meijide et al. (2009)	47, 48, 49, 50	51	0.95	-0.06	0.03	0.91	-0.10	0.15
Sánchez-Martín et al. (2010)	63, 64	31	0.99	-0.01	0.01	0.98	-0.02	0.01
Sánchez-Martín et al. (2010)	67, 68	37	1.00	-0.01	0.01	0.99	-0.01	0.03
Vallejo et al. (2006)	79, 80, 81	83	0.76	-0.27	0.00	0.51	-0.67	0.02

Appendix A2. Supplementary data of Study 2

Table A2.1 Basic information, soil organic carbon (SOC) content and its response ratio (RR) of all studied datasets. Only the datasets with words in the column "Aggregated" were included in the meta-analysis (see Methods). The datasets having more than one code number are aggregated datasets composed of the datasets to which the numbers refer. ORG: organic management; LT: land treatment (urban wastes and C inputs exceeding 10 Mg C ha−1); OA: organic amendments; CC: cover crops; Slurry: liquid manures; NT: no tillage; RT: reduced tillage; CMP: combined management practices (OA combined with CC, CR, RT or NT); Unfertilized: no organic or synthetic fertilizers are applied.

			Category		SOC (g C/kg)			
Code	Reference	Crop type	Total	Aggregated	Initial	Final	Conventional	RR
1	Agenbag and Maree (1989)	Wheat	NT	NT	13.3	14.1	10.7	1.32
2	Agenbag and Maree (1989)	Wheat-pasture	NT	NT	15.1	20.2	6.8	2.97
3	Agenbag and Maree (1989)	Wheat	RT	RT	13.3	13.3	10.7	1.24
4	Agenbag and Maree (1989)	Wheat-pasture	RT	RT	15.1	15.1	6.8	2.22
6-13	Albiach et al. (2001)	Lettuce-chard		LT	10.6	15.5	11.8	1.32
14, 15	Albiach et al. (2001)	Lettuce-chard		OA	7.3	10.5	8.4	1.25
12, 13	Albiach et al. (2001)	Lettuce-chard		LT	7.3	13.0	8.4	1.55
5	Albiach et al. (2001)	Lettuce-chard	Control	Control	10.6	10.6	11.8	0.90
6	Albiach et al. (2001)	Lettuce-chard	LT		10.6	12.2	11.8	1.04
7	Albiach et al. (2001)	Lettuce-chard	LT		10.6	15.3	11.8	1.30
8	Albiach et al. (2001)	Lettuce-chard	LT		10.6	18.1	11.8	1.54
9	Albiach et al. (2001)	Lettuce-chard	LT		10.6	13.9	11.8	1.18
10	Albiach et al. (2001)	Lettuce-chard	LT		10.6	16.0	11.8	1.35
11	Albiach et al. (2001)	Lettuce-chard	LT		10.6	17.5	11.8	1.49
12	Albiach et al. (2001)	Lettuce-chard	LT		7.3	14.6	8.4	1.74
13	Albiach et al. (2001)	Lettuce-chard	LT		7.3	11.4	8.4	1.36
14	Albiach et al. (2001)	Lettuce-chard	OA		7.3	8.0	8.4	0.95
15	Albiach et al. (2001)	Lettuce-chard	OA		7.3	13.1	8.4	1.56
16, 17	Altieri et al. (2008)	Olive		OA	14.2	17.3	14.7	1.18
16	Altieri et al. (2008)	Olive	OA		14.2	18.5	14.7	1.26

17	Altieri et al. (2008)	Olive	OA		14.2	16.1	14.7	1.10
20, 21	Alvaro Fuentes et al. (2008)	Wheat-Barley-Rapeseed		RT				0.97
22, 23	Alvaro Fuentes et al. (2008)	Wheat-Barley		RT				0.97
18	Alvaro Fuentes et al. (2008)	Wheat-Barley-Rapeseed	NT	NT				0.88
19	Alvaro Fuentes et al. (2008)	Wheat-Barley	NT	NT				1.01
20	Alvaro Fuentes et al. (2008)	Wheat-Barley-Rapeseed	RT					0.98
21	Alvaro Fuentes et al. (2008)	Wheat-Barley-Rapeseed	RT					0.97
22	Alvaro Fuentes et al. (2008)	Wheat-Barley	RT					0.95
23	Alvaro Fuentes et al. (2008)	Wheat-Barley	RT					0.99
24	Alvaro Fuentes et al. (2009)	Barley	NT	NT				1.23
25	Alvaro Fuentes et al. (2009)	Barley-Fallow	NT	NT				1.08
26	Alvaro Fuentes et al. (2009)	Barley	RT	RT				1.12
27	Alvaro Fuentes et al. (2009)	Barley-Fallow	RT	RT				1.01
30, 32	Andrews et al. (2002)	Vegetables		OA, ORG		11.4	7.8	1.46
28	Andrews et al. (2002)	Vegetables, cotton	CC	CC	12.0	9.6	7.3	1.32
29	Andrews et al. (2002)	Vegetables, cotton	OA	OA	10.7	11.1	8.7	1.28
30	Andrews et al. (2002)	Vegetables	OA, ORG			9.9	7.8	1.28
31	Andrews et al. (2002)	Vegetables	OA, ORG			10.6	7.8	1.36
32	Andrews et al. (2002)	Vegetables	OA, ORG			13.6	7.8	1.75
33	Andrews et al. (2002)	Vegetables, cotton	RMP	RMP	11.0	12.4	11.6	1.07
34	Andrews et al. (2002)	Vegetables, cotton	RMP	RMP	10.7	8.8	8.6	1.02
35	Andrews et al. (2002)	Vegetables, cotton	RMP	RMP	10.1	12.2	9.0	1.36
36	Andrews et al. (2002)	Vegetables, cotton	RMP	RMP	6.4	5.7	3.5	1.63
37, 38, 39	Antoniadis et al. (2010)	Cotton		LT		19.3	13.7	1.40
37	Antoniadis et al. (2010)	Cotton	LT			16.6	13.7	1.21
38	Antoniadis et al. (2010)	Cotton	LT			19.5	13.7	1.42
39	Antoniadis et al. (2010)	Cotton	LT			21.8	13.7	1.58
40	Aranda et al. (2011)	Olive	RMP, ORG	RMP, ORG		13.1	7.9	1.67
41	Aranda et al. (2011)	Olive	RMP, ORG	RMP, ORG		12.2	5.9	2.09
42	Badalucco et al. (2010)	Vegetables	RMP	RMP	10.1	19.3	10.9	1.77
45-47	Benitez et al. (2006)	Olive		ORG		10.4	7.9	1.44
50-53	Benitez et al. (2006)	Olive		CC, ORG		8.8	7.9	1.21
48, 49	Benitez et al. (2006)	Olive		ORG		12.7	13.0	1.01

54, 55	Benitez et al. (2006)	Olive		CC, ORG		12.6	13.0	1.00
43	Benitez et al. (2006)	Olive	NT	NT		10.7	15.3	0.70
44	Benitez et al. (2006)	Olive	ORG			13.2	10.3	1.28
45	Benitez et al. (2006)	Olive	ORG			10.8	10.3	1.05
46	Benitez et al. (2006)	Olive	ORG			13.2	5.6	2.38
47	Benitez et al. (2006)	Olive	ORG			10.8	5.6	1.94
48	Benitez et al. (2006)	Olive	ORG			12.7	10.7	1.19
49	Benitez et al. (2006)	Olive	ORG			12.7	15.3	0.83
50	Benitez et al. (2006)	Olive	CC, ORG			11.0	10.3	1.06
51	Benitez et al. (2006)	Olive	CC, ORG			6.5	10.3	0.63
52	Benitez et al. (2006)	Olive	CC, ORG			11.0	5.6	1.97
53	Benitez et al. (2006)	Olive	CC, ORG			6.5	5.6	1.18
54	Benitez et al. (2006)	Olive	CC, ORG			12.6	10.7	1.18
55	Benitez et al. (2006)	Olive	CC, ORG			12.6	15.3	0.82
56	Bessam and Mrabet (2003)	Wheat rotations	NT	NT	11.0			1.12
57	Campos-Herrera et al. (2010)	Arable crops	ORG	ORG		15.4	14.8	1.04
58	Campos-Herrera et al. (2010)	Orchards	ORG	ORG		17.4	17.5	0.99
59	Campos-Herrera et al. (2010)	Vineyards	ORG	ORG		11.8	9.3	1.26
60	Canali et al. (2003)	Citrus	ORG	ORG		13.3	10.8	1.24
61-63	Canali et al. (2004)	Citrus		OA, ORG		19.4	17.3	1.12
61	Canali et al. (2004)	Citrus	OA, ORG			18.4	17.3	1.06
62	Canali et al. (2004)	Citrus	OA, ORG			20.5	17.3	1.18
63	Canali et al. (2004)	Citrus	OA, ORG			19.3	17.3	1.12
64	Canali et al. (2009)	Citrus	RMP, ORG	RMP, ORG		10.9	8.8	1.24
65, 66	Carbonell-Bojollo et al. (2009)	Olive		OA				1.13
65	Carbonell-Bojollo et al. (2009)	Olive	OA					1.16
66	Carbonell-Bojollo et al. (2009)	Olive	OA					1.10
67	Castro et al. (2008)	Olive	CC	CC	5.0	7.3	9.4	0.77
68	Castro et al. (2008)	Olive	CC	CC	5.0	10.7	9.4	1.14
69	Castro et al. (2008)	Olive	NT	NT	5.0	5.0	9.4	0.53
70	Celik et al. (2004)	Wheat-pepper-maize	Control	Control	8.7	9.4	9.6	0.98
71	Celik et al. (2004)	Wheat-pepper-maize	OA	OA	8.7	10.1	9.6	1.06
72	Celik et al. (2004)	Wheat-pepper-maize	OA	OA	8.7	10.1	9.6	1.05

73	Celik et al. (2004)	Wheat-pepper-maize	OA	OA	8.7	9.4	9.6	0.98
75-77	Clark et al. (1998)	Tomato-safflower-beans-maize		ORG		10.2	9.4	1.08
76, 77	Clark et al. (1998)	Tomato-safflower-beans-maize		RMP		10.2	9.3	1.10
74	Clark et al. (1998)	Tomato-safflower-beans-maize	CC	CC		9.7	9.1	1.07
75	Clark et al. (1998)	Tomato-safflower-beans-maize	OA, ORG	OA		10.2	9.7	1.05
76	Clark et al. (1998)	Tomato-safflower-beans-maize	RMP, ORG			10.2	9.1	1.12
77	Clark et al. (1998)	Tomato-safflower-beans-maize	RMP, ORG			10.2	9.4	1.08
78, 79	Cookson et al. (2006)	Wheat rotations		OA, ORG		18.8	14.9	1.26
78	Cookson et al. (2006)	Wheat rotations	OA, ORG			18.5	14.9	1.24
79	Cookson et al. (2006)	Wheat rotations	OA, ORG			19.0	14.9	1.27
80	Deria et al. (2003)	Wheat rotations	Control, ORG	Control, ORG		4.0	5.4	0.74
81	Deria et al. (2003)	Wheat rotations	Control, ORG	Control, ORG		5.5	5.3	1.04
82	Deria et al. (2003)	Wheat rotations	Control, ORG	Control, ORG		5.5	6.8	0.81
83	Deria et al. (2003)	Wheat rotations	Control, ORG	Control, ORG		7.5	6.9	1.09
84	Deria et al. (2003)	Wheat rotations	OA, ORG	OA, ORG		9.1	9.5	0.96
85	Deria et al. (2003)	Wheat rotations	OA, ORG	OA, ORG		19.0	17.4	1.09
86	Deria et al. (2003)	Wheat rotations	OA, ORG	OA, ORG		1.4	1.6	0.88
87	Drinkwater et al. (1995)	Wheat	RMP, ORG	RMP, ORG		12.5	9.8	1.28
88, 89	Efthimiadou et al. (2010)	Beans		OA, ORG		14.1	14.7	0.96
88	Efthimiadou et al. (2010)	Beans	OA, ORG			12.0	14.7	0.81
89	Efthimiadou et al. (2010)	Beans	OA, ORG			16.3	14.7	1.11
90-97	Fernández et al. (2009)	Barley		LT	7.2	14.7	5.8	2.53
90	Fernández et al. (2009)	Barley	LT		7.2	17.0	5.8	2.93
91	Fernández et al. (2009)	Barley	LT		7.2	31.0	5.8	5.34
92	Fernández et al. (2009)	Barley	LT		7.2	15.2	5.8	2.62
93	Fernández et al. (2009)	Barley	LT		7.2	20.3	5.8	3.50

94	Fernández et al. (2009)	Barley	LT		7.2	6.2	5.8	1.07
95	Fernández et al. (2009)	Barley	LT		7.2	7.3	5.8	1.26
96	Fernández et al. (2009)	Barley	LT		7.2	7.9	5.8	1.36
97	Fernández et al. (2009)	Barley	LT		7.2	12.3	5.8	2.12
98	Fernández-Ugalde et al. (2009)	Barley	NT	NT		10.0	9.1	1.09
100, 101	Gacía-Gil et al. (2000)	Barley		LT	8.0	18.2	6.8	2.68
99	Gacía-Gil et al. (2000)	Barley	Control	Control	8.0	5.9	6.8	0.86
100	Gacía-Gil et al. (2000)	Barley	LT		8.0	11.3	6.8	1.66
101	Gacía-Gil et al. (2000)	Barley	LT		8.0	25.1	6.8	3.69
102	Gacía-Gil et al. (2000)	Barley	OA	OA	8.0	9.8	6.8	1.44
103	García-Ruiz et al. (2009)	Olive	OA, ORG	OA, ORG		28.9	21.0	1.38
104	García-Ruiz et al. (2009)	Olive	RMP, ORG	RMP, ORG		47.3	30.9	1.53
105	García-Ruiz et al. (2009)	Olive	RMP, ORG	RMP, ORG		29.1	19.3	1.51
106	Gómez et al. (1999)	Olive	NT	NT		8.7	8.2	1.06
107	Gómez et al. (2009)	Olive	NT	NT	6.2	5.3	7.4	0.72
108	Gómez et al. (2009)	Olive	RMP	RMP	6.2	9.3	7.4	1.26
109	Herencia et al. (2007)	Vegetables	RMP, ORG	RMP, ORG	7.6	18.5	7.3	2.54
110	Herencia et al. (2007)	Vegetables	RMP, ORG	RMP, ORG	7.6	18.8	8.0	2.33
111	Herencia et al. (2007)	Vegetables	RMP, ORG	RMP, ORG	7.6	17.0	8.4	2.02
112	Herencia et al. (2011)	Vegetables	RMP, ORG	RMP, ORG	6.3	15.8	7.3	2.16
113	Herencia et al. (2011)	Vegetables	RMP, ORG	RMP, ORG	6.3	14.3	5.8	2.47
114	Herencia et al. (2011)	Vegetables	RMP, ORG	RMP, ORG	6.3	13.5	5.5	2.47
115-117	Hernández et al. (2005)	Olive		CC	2.4	2.8	1.9	1.49
115	Hernández et al. (2005)	Olive	CC		2.6	3.3	1.9	1.74
116	Hernández et al. (2005)	Olive	CC		2.5	3.4	1.9	1.79
117	Hernández et al. (2005)	Olive	CC		2.1	1.8	1.9	0.95
118	Hernández et al. (2005)	Olive	NT	NT	2.8	1.7	1.9	0.89
119	Hernanz et al. (2009)	Wheat-legume	NT	NT	5.3	7.1	6.1	1.16
120	Hernanz et al. (2009)	Wheat-legume	RT	RT	5.3	6.1	6.1	1.00
121-125	Hojati et al. (2006)	Maize		LT	4.9	26.8	5.1	5.26
121	Hojati et al. (2006)	Maize	Control	Control	4.9	4.9	5.1	0.96
122	Hojati et al. (2006)	Maize	LT		4.9	22.7	5.1	4.45
123	Hojati et al. (2006)	Maize	LT		4.9	22.1	5.1	4.33

124	Hojati et al. (2006)	Maize	LT		4.9	18.6	5.1	3.65
125	Hojati et al. (2006)	Maize	LT		4.9	43.9	5.1	8.61
127-130	Kong et al. (2005)	Wheat rotations		CC		9.5	8.7	1.09
134-136	Kong et al. (2005)	Maize rotations		ORG		12.2	9.2	1.32
135, 136	Kong et al. (2005)	Maize rotations		RMP		12.2	9.3	1.31
126	Kong et al. (2005)	Wheat rotations	CC	CC		10.5	9.5	1.09
127	Kong et al. (2005)	Wheat rotations	CC			9.5	8.5	1.12
128	Kong et al. (2005)	Wheat rotations	CC			9.5	8.6	1.11
129	Kong et al. (2005)	Wheat rotations	CC			9.5	9.1	1.04
130	Kong et al. (2005)	Wheat rotations	CC			9.0	9.4	0.95
131	Kong et al. (2005)	Maize rotations	CC	CC		9.0	9.1	0.99
132	Kong et al. (2005)	Wheat rotations	Control	Control		9.0	9.5	0.94
133	Kong et al. (2005)	Wheat rotations	Control	Control		8.5	8.6	0.99
134	Kong et al. (2005)	Maize rotations	OA, ORG	OA		12.2	9.0	1.35
135	Kong et al. (2005)	Maize rotations	RMP, ORG			12.2	9.4	1.29
136	Kong et al. (2005)	Maize rotations	RMP, ORG			12.2	9.1	1.33
137	Laudicina et al. (2010)	Vegetables	OA	OA		10.5	6.4	1.64
138	Laudicina et al. (2010)	Vegetables	RMP	RMP		16.3	6.4	2.55
139	Laudicina et al. (2010)	Vegetables	RT	RT		8.3	6.4	1.30
142, 143	Lithourgidis et al. (2007)	Maize		Slurry	5.6	5.7	5.5	1.05
140, 141	Lithourgidis et al. (2007)	Maize		Control	5.2	5.3	5.5	0.97
140	Lithourgidis et al. (2007)	Maize	Control		5.2	5.3	5.5	0.96
141	Lithourgidis et al. (2007)	Maize	Control		5.2	5.3	5.4	0.98
142	Lithourgidis et al. (2007)	Maize	Slurry		5.6	5.7	5.4	1.06
143	Lithourgidis et al. (2007)	Maize	Slurry		5.6	5.7	5.5	1.04
144, 145	López et al. (1996)	Olive		LT	4.5	11.8	4.5	2.62
144	López et al. (1996)	Olive	LT		4.5	10.6	4.5	2.34
145	López et al. (1996)	Olive	LT		4.5	13.1	4.5	2.90
146	López-Bellido et al. (2010)	Wheat rotations	NT	NT				1.05
147	López-Bellido et al. (2010)	Wheat rotations	NT	NT				1.06
148	López-Bellido et al. (2010)	Wheat rotations	NT	NT				1.36
149	López-Bellido et al. (2010)	Wheat rotations	NT	NT				1.00
150	López-Bellido et al. (2010)	Wheat rotations	NT	NT				1.55

151	López-Garrido et al. (2009)	Wheat rotations	NT	NT	7.4	8.2	7.4	1.11
152	Lopez-Piñeiro et al. (2008)	Olive	LT	LT	13.3	33.5	11.9	2.82
153	Lopez-Piñeiro et al. (2008)	Olive	OA	OA	13.3	19.5	11.9	1.64
154, 155	Lopez-Piñeiro et al. (2010)	Olive		LT	13.3	40.8	10.3	3.98
154	Lopez-Piñeiro et al. (2010)	Olive	LT		13.3	33.5	10.3	3.27
155	Lopez-Piñeiro et al. (2010)	Olive	LT		13.3	48.1	10.3	4.69
156-158	Madejón et al. (2003)	Orange		LT	4.2	9.2	4.6	2.00
156	Madejón et al. (2003)	Orange	LT		4.2	9.7	4.6	2.09
157	Madejón et al. (2003)	Orange	LT		4.2	8.8	4.6	1.90
158	Madejón et al. (2003)	Orange	OA	OA	4.2	7.9	4.6	1.70
159	Marinari et al. (2010a)	Wheat rotations	OA, ORG	OA, ORG		29.8	27.0	1.10
160	Marinari et al. (2010a)	Wheat rotations	OA, ORG	OA, ORG		11.3	7.6	1.49
161	Marinari et al. (2010b)	Wheat rotations	RMP, ORG	RMP, ORG	11.7	12.8	10.9	1.17
162	Martínez et al. (2008)	Wheat-maize	NT	NT	11.0	23.2	22.2	1.05
163	Martín-Rueda et al. (2007)	Barley rotations	NT	NT				1.69
164	Martín-Rueda et al. (2007)	Barley rotations	RT	RT				1.48
167, 168	Matsi et al. (2003)	Wheat		Slurry	7.5	7.2	7.0	1.05
165, 166	Matsi et al. (2003)	Wheat		Control	7.5	5.6	7.0	0.81
165	Matsi et al. (2003)	Wheat	Control		7.5	5.6	6.1	0.92
166	Matsi et al. (2003)	Wheat	Control		7.5	5.6	7.9	0.71
167	Matsi et al. (2003)	Wheat	Slurry		7.5	7.2	6.1	1.18
168	Matsi et al. (2003)	Wheat	Slurry		7.5	7.2	7.9	0.91
169	Mazzoncini et al. (2010)	Wheat rotations	CC, ORG	CC, ORG	9.4	9.5	7.8	1.22
170-172	Mazzoncini et al. (2011)	Wheat rotations		CC		11.5	10.8	1.06
170	Mazzoncini et al. (2011)	Wheat rotations	CC			11.0	10.8	1.02
171	Mazzoncini et al. (2011)	Wheat rotations	CC			11.6	10.8	1.07
172	Mazzoncini et al. (2011)	Wheat rotations	CC			11.9	10.8	1.10
173	Mazzoncini et al. (2011)	Wheat rotations	NT	NT		12.0	10.7	1.12
174	Melero et al. (2006)	Vegetables	RMP, ORG	RMP, ORG	8.1	20.5	8.7	2.36
175, 176	Melero et al. (2007)	Wheat rotations		OA, ORG	9.8	22.8	13.2	1.73
175	Melero et al. (2007)	Wheat rotations	OA, ORG		9.8	22.1	13.2	1.67
176	Melero et al. (2007)	Wheat rotations	OA, ORG		9.8	23.5	13.2	1.78
177, 178	Melero et al. (2008)	Vegetables		OA, ORG	7.6	13.8	8.5	1.62

177	Melero et al. (2008)	Vegetables	OA, ORG		7.6	13.5	8.5	1.59
178	Melero et al. (2008)	Vegetables	OA, ORG		7.6	14.0	8.5	1.65
179	Melero et al. (2009)	Wheat rotations	NT	NT		9.4	7.3	1.29
180	Melero et al. (2009)	Wheat rotations	RT	RT		9.3	8.8	1.05
181-183	Monokrousos et al. (2006)	Asparragus		OA, ORG		14.6	15.2	0.96
181	Monokrousos et al. (2006)	Asparragus	OA, ORG			17.2	15.2	1.13
182	Monokrousos et al. (2006)	Asparragus	OA, ORG			13.4	15.2	0.88
183	Monokrousos et al. (2006)	Asparragus	OA, ORG			13.1	15.2	0.86
184	Montanaro et al. (2009)	Kiwifruit	RMP	RMP	8.3	9.0	8.7	1.03
185	Montanaro et al. (2009)	Apricot	RMP	RMP	9.1	9.2	9.5	0.97
187-189	Montemurro et al. (2010a)	Lettuce		OA	21.4	23.5	21.9	1.07
186	Montemurro et al. (2010a)	Lettuce	Control	Control	21.4	20.1	21.9	0.92
187	Montemurro et al. (2010a)	Lettuce	OA		21.4	23.2	21.9	1.06
188	Montemurro et al. (2010a)	Lettuce	OA		21.4	21.9	21.9	1.00
189	Montemurro et al. (2010a)	Lettuce	OA		21.4	25.5	21.9	1.16
190	Montemurro et al. (2010a)	Lettuce	OA	OA	21.4	20.4	21.9	0.93
191, 192	Montemurro et al. (2010b)	Alfalfa		LT	12.5	21.4	17.0	1.26
193, 194	Montemurro et al. (2010b)	Cocksfoot		LT	12.4	18.0	14.2	1.27
191	Montemurro et al. (2010b)	Alfalfa	LT		12.5	21.3	17.0	1.25
192	Montemurro et al. (2010b)	Alfalfa	LT		12.5	21.6	17.0	1.27
193	Montemurro et al. (2010b)	Cocksfoot	LT		12.4	19.0	14.2	1.34
194	Montemurro et al. (2010b)	Cocksfoot	LT		12.4	17.0	14.2	1.20
195	Moreno et al. (2006)	Wheat-sunflower	RT	RT	7.7	7.7	7.6	1.01
197-201	Morra et al. (2010)	Vegetables		LT	26.0	28.2	23.9	1.18
196	Morra et al. (2010)	Vegetables	Control	Control	26.0	24.4	23.9	1.02
197	Morra et al. (2010)	Vegetables	LT		26.0	25.2	23.9	1.05
198	Morra et al. (2010)	Vegetables	LT		26.0	29.7	23.9	1.24
199	Morra et al. (2010)	Vegetables	LT		26.0	32.2	23.9	1.35
200	Morra et al. (2010)	Vegetables	LT		26.0	27.2	23.9	1.14
201	Morra et al. (2010)	Vegetables	LT		26.0	26.5	23.9	1.11
202	Moussa Machraoui et al. (2010)	Wheat	NT	NT	13.0	15.0	13.0	1.15
203	Moussa Machraoui et al. (2010)	Barley	NT	NT	15.0	17.0	15.0	1.13

204	Moussa Machraoui et al. (2010)	Wheat	NT	NT	9.0	11.0	9.0	1.22
205	Moussa Machraoui et al. (2010)	Pea	NT	NT	8.0	9.0	8.0	1.13
206	Moussa Machraoui et al. (2010)	Oats	NT	NT	10.0	11.0	10.0	1.10
207	Mrabet et al. (2001)	Wheat rotations	NT	NT				1.10
208-210	Muñoz et al. (2007)	Maize		RMP	7.2	9.8	7.2	1.37
208	Muñoz et al. (2007)	Maize	NT	NT	7.2	8.3	7.2	1.15
209	Muñoz et al. (2007)	Maize	RMP		7.2	7.8	7.2	1.09
210	Muñoz et al. (2007)	Maize	RMP		7.2	11.8	7.2	1.64
211	Nieto et al. (2010)	Olive	LT	LT		36.1	6.7	5.37
212	Nieto et al. (2010)	Olive	LT	LT		28.7	7.1	4.03
214, 215	Okur et al. (2009)	Vineyard		ORG	7.5	9.5	8.0	1.18
213-215	Okur et al. (2009)	Vineyard		RMP	7.5	9.7	8.0	1.21
213	Okur et al. (2009)	Vineyard	CC, ORG	CC	7.5	9.0	8.0	1.13
214	Okur et al. (2009)	Vineyard	RMP, ORG		7.5	9.8	8.0	1.23
215	Okur et al. (2009)	Vineyard	RMP, ORG		7.5	9.6	8.0	1.20
216	Pardo et al. (2009)	Wheat-fallow-barley-vetch	Control	Control	12.8	12.5	13.9	0.89
217	Pardo et al. (2009)	Wheat-fallow-barley-vetch	Control	Control	13.9	12.6	12.1	1.04
218	Pardo et al. (2009)	Wheat-fallow-barley-vetch	OA, ORG	OA, ORG	12.8	13.8	13.9	0.99
219	Pardo et al. (2009)	Wheat-fallow-barley-vetch	OA, ORG	OA, ORG	13.9	12.8	12.1	1.06
220-224	Plaza et al. (2004)	Barley		Slurry		13.0	13.1	1.00
220	Plaza et al. (2004)	Barley	Control	Control		13.2	13.1	1.01
221	Plaza et al. (2004)	Barley	Slurry			13.3	13.1	1.02
222	Plaza et al. (2004)	Barley	Slurry			13.4	13.1	1.02
223	Plaza et al. (2004)	Barley	Slurry			13.2	13.1	1.01
224	Plaza et al. (2004)	Barley	Slurry			13.0	13.1	0.99
225	Plaza et al. (2004)	Barley	Slurry			12.3	13.1	0.94
226, 227	Raviv et al. (2006)	Stone fruits		ORG		18.0	11.0	1.64
226	Raviv et al. (2006)	Stone fruits	OA, ORG	OA		15.0	11.0	1.36
227	Raviv et al. (2006)	Stone fruits	RMP, ORG	RMP		21.0	11.0	1.91
228	Reganold et al. (2010)	Strawberry	RMP, ORG	RMP, ORG		9.6	7.9	1.22
229	Romanya and Rovira (2009)	Barley-fallow	OA, ORG	OA, ORG	8.1	7.2	8.1	0.89

230	Romanya and Rovira (2009)	Wheat-pea	OA, ORG	OA, ORG	7.0	9.2	7.0	1.32
231-237	Ryan et al. (2008)	Wheat rotations		RT	6.4	7.2	6.4	1.12
231	Ryan et al. (2008)	Wheat-pea	RT		6.4	6.5	6.4	1.02
232	Ryan et al. (2008)	Wheat-fallow	RT		6.4	6.7	6.4	1.05
233	Ryan et al. (2008)	Wheat monoculture	RT		6.4	7.3	6.4	1.14
234	Ryan et al. (2008)	Wheat-lentil	RT		6.4	6.7	6.4	1.05
235	Ryan et al. (2008)	Wheat-chickpea	RT		6.4	7.4	6.4	1.15
236	Ryan et al. (2008)	Wheat-vetch	RT		6.4	7.4	6.4	1.16
237	Ryan et al. (2008)	Wheat-medicago	RT		6.4	8.1	6.4	1.26
238	Sombrero et al. (2010)	Wheat rotations	NT	NT				1.33
239	Sombrero et al. (2010)	Wheat rotations	RT	RT				1.21
240-243	Sommer et al. (2011)	Cereal rotations		OA		11.9	10.1	1.17
244-251	Sommer et al. (2011)	Cereal rotations		RMP		11.9	9.2	1.29
240	Sommer et al. (2011)	Cereal rotations	OA			13.8	10.4	1.32
241	Sommer et al. (2011)	Cereal rotations	OA			11.7	10.4	1.12
242	Sommer et al. (2011)	Cereal rotations	OA			12.6	9.8	1.29
243	Sommer et al. (2011)	Cereal rotations	OA			9.3	9.8	0.95
244	Sommer et al. (2011)	Cereal rotations	RMP			13.8	9.3	1.49
245	Sommer et al. (2011)	Cereal rotations	RMP			11.7	9.3	1.27
246	Sommer et al. (2011)	Cereal rotations	RMP			13.8	9.1	1.52
247	Sommer et al. (2011)	Cereal rotations	RMP			11.7	9.1	1.29
248	Sommer et al. (2011)	Cereal rotations	RMP			12.6	9.3	1.36
249	Sommer et al. (2011)	Cereal rotations	RMP			9.3	9.3	1.00
250	Sommer et al. (2011)	Cereal rotations	RMP			12.6	9.1	1.38
251	Sommer et al. (2011)	Cereal rotations	RMP			9.3	9.1	1.02
252	Sommer et al. (2011)	Cereal rotations	RT	RT		10.5	9.3	1.13
253	Sommer et al. (2011)	Cereal rotations	RT	RT		10.1	9.1	1.11
254	Sommer et al. (2011)	Cereal rotations	RT	RT		10.4	9.8	1.07
255-260	Tejada et al. (2006)	Wheat		OA	7.3	16.8	6.6	2.56
255	Tejada et al. (2006)	Wheat	OA		7.3	14.3	6.6	2.18
256	Tejada et al. (2006)	Wheat	OA		7.3	16.2	6.6	2.48
257	Tejada et al. (2006)	Wheat	OA		7.3	18.9	6.6	2.88
258	Tejada et al. (2006)	Wheat	OA		7.3	14.9	6.6	2.28

259	Tejada et al. (2006)	Wheat	OA		7.3	16.9	6.6	2.58
260	Tejada et al. (2006)	Wheat	OA		7.3	19.6	6.6	3.00
261	Vavoulidou et al. (2006)	Vineyard	ORG	ORG		5.6	3.9	1.41
262	Vavoulidou et al. (2009)	Vineyard	ORG	ORG		9.1	6.8	1.34
263	Vavoulidou et al. (2009)	Olive	ORG	ORG		17.7	24.2	0.73
264	Vavoulidou et al. (2009)	Vineyard	ORG	ORG		7.7	8.0	0.97
265	Vavoulidou et al. (2009)	Olive	ORG	ORG		13.9	17.5	0.79
266	Vavoulidou et al. (2009)	Citrus	ORG	ORG		16.8	16.2	1.04
267, 268	Veenstra et al. (2007)	Cotton-tomato		CC	6.3	7.2	6.2	1.16
267	Veenstra et al. (2007)	Cotton-tomato	CC		6.3	7.0	6.2	1.14
268	Veenstra et al. (2007)	Cotton-tomato	CC		6.3	7.3	6.2	1.18
269	Veenstra et al. (2007)	Cotton-tomato	RT	RT	6.3	6.6	6.2	1.07
270, 271	Virto et al. (2007)	Barley		NT	5.9	10.2	9.7	1.05
270	Virto et al. (2007)	Barley	NT		5.9	10.2	9.7	1.05
271	Virto et al. (2007)	Barley	NT		5.9	10.2	9.7	1.05

Table A2.2 C sequestration rate and C input rate of all studied datasets. Only the datasets with words in the column "Aggregated" were included in the meta-analysis (see Methods). The datasets having more than one code number are aggregated datasets composed of the datasets to which the numbers refer. Net C sequestration rate and net C input rate are calculated subtracting Conventional values from Treatment values. ORG: organic management; LT: land treatment (urban wastes and C inputs exceeding 10 Mg C ha−1); OA: organic amendments; CC: cover crops; Slurry: liquid manures; NT: no tillage; RT: reduced tillage; CMP: combined management practices (OA combined with CC, CR, RT or NT); Unfertilized: no organic or synthetic fertilizers are applied; CR: Crop residues; MSW: Municipal solid waste; SS: Sewage sludge; OMW: Olive mill waste; AIW: Agro-industry waste

		Category		C sequestration rate (Mg C/ha/yr)			C input (Mg C/ha/yr)		
Code	Reference	Total	Aggregated	Treatment	Convent.	Net	Type	Treat.	Net
1	Agenbag and Maree (1989)	NT	NT	0.22	0.00	0.22	CR		
2	Agenbag and Maree (1989)	NT	NT	0.86	0.00	0.86	CR		
3	Agenbag and Maree (1989)	RT	RT	0.17	0.00	0.17	CR		
4	Agenbag and Maree (1989)	RT	RT	0.55	0.00	0.55	CR		
6-13	Albiach et al. (2001)		LT	1.60	0.39	1.21	MSW/SS	8.43	8.43
14, 15	Albiach et al. (2001)		OA	1.18	0.41	0.77	compost and	3.50	3.50

							manure		
12, 13	Albiach et al. (2001)		LT	2.07	0.41	1.65	MSW/SS	5.76	5.76
5	Albiach et al. (2001)	Control	Control	0.00	0.39	-0.39	none specified		
6	Albiach et al. (2001)	LT		0.55	0.39	0.16	MSW/SS	4.27	4.27
7	Albiach et al. (2001)	LT		1.54	0.39	1.14	MSW/SS	8.54	8.54
8	Albiach et al. (2001)	LT		2.43	0.39	2.04	MSW/SS	12.81	12.81
9	Albiach et al. (2001)	LT		1.10	0.39	0.71	MSW/SS	4.16	4.16
10	Albiach et al. (2001)	LT		1.76	0.39	1.36	MSW/SS	8.33	8.33
11	Albiach et al. (2001)	LT		2.25	0.39	1.86	MSW/SS	12.49	12.49
12	Albiach et al. (2001)	LT		2.62	0.41	2.21	MSW/SS	6.81	6.81
13	Albiach et al. (2001)	LT		1.52	0.41	1.10	MSW/SS	4.72	4.72
14	Albiach et al. (2001)	OA		0.26	0.41	-0.15	compost	0.56	0.56
15	Albiach et al. (2001)	OA		2.10	0.41	1.69	manure	6.44	6.44
16, 17	Altieri et al. (2008)		OA	1.38	0.22	1.17		3.74	3.74
16	Altieri et al. (2008)	OA		1.91	0.22	1.69	compost OMW	3.55	3.55
17	Altieri et al. (2008)	OA		0.86	0.22	0.64	compost OMW	3.94	3.94
20, 21	Alvaro Fuentes et al. (2008)		RT	-0.10	0.00	-0.10			
22, 23	Alvaro Fuentes et al. (2008)		RT	-0.09	0.00	-0.09			
18	Alvaro Fuentes et al. (2008)	NT	NT	-0.43	0.00	-0.43	CR		
19	Alvaro Fuentes et al. (2008)	NT	NT	0.02	0.00	0.02	CR		
20	Alvaro Fuentes et al. (2008)	RT		-0.08	0.00	-0.08	CR		
21	Alvaro Fuentes et al. (2008)	RT		-0.12	0.00	-0.12	CR		
22	Alvaro Fuentes et al. (2008)	RT		-0.16	0.00	-0.16	CR		
23	Alvaro Fuentes et al. (2008)	RT		-0.02	0.00	-0.02	CR		
24	Alvaro Fuentes et al. (2009)	NT	NT	0.46	0.18	0.29	CR	1.09	-0.22
25	Alvaro Fuentes et al. (2009)	NT	NT	0.15	-0.01	0.16	CR	0.33	-0.16
26	Alvaro Fuentes et al. (2009)	RT	RT	0.24	0.18	0.06	CR	1.14	-0.16
27	Alvaro Fuentes et al. (2009)	RT	RT	0.01	-0.01	0.01	CR	0.47	-0.02
30, 32	Andrews et al. (2002)		OA, ORG	1.43	0.00	1.43	manure, compost		
28	Andrews et al. (2002)	CC	CC	-1.80	-2.24	0.44	CC		
29	Andrews et al. (2002)	OA	OA	0.24	-1.43	1.67	C + M		
30	Andrews et al. (2002)	OA, ORG		1.43	0.00	1.43	manure		
31	Andrews et al. (2002)	OA, ORG		1.90	0.00	1.90	compost		

32	Andrews et al. (2002)	OA, ORG		0.96	0.00	0.96	manure		
33	Andrews et al. (2002)	RMP	RMP	0.92	0.50	0.42	C + CC		
34	Andrews et al. (2002)	RMP	RMP	-0.99	0.19	-1.18	C + CC		
35	Andrews et al. (2002)	RMP	RMP	1.28	0.56	0.72	C + M + CC		
36	Andrews et al. (2002)	RMP	RMP	-0.46	-1.93	1.47	C + M + CC		
37, 38, 39	Antoniadis et al. (2010)		LT	3.50	0.00	3.50	MSW/SS	7.48	7.48
37	Antoniadis et al. (2010)	LT		1.57	0.00	1.57	MSW/SS	2.49	2.49
38	Antoniadis et al. (2010)	LT		3.77	0.00	3.77	MSW/SS	7.48	7.48
39	Antoniadis et al. (2010)	LT		5.15	0.00	5.15	MSW/SS	12.47	12.47
40	Aranda et al. (2011)	RMP, ORG	RMP, ORG	0.60	0.00	0.60	manure + CC		
41	Aranda et al. (2011)	RMP, ORG	RMP, ORG	0.42	0.00	0.42	manure + CC		
42	Badalucco et al. (2010)	RMP	RMP	1.35	0.20	1.15	compost		
45-47	Benitez et al. (2006)		ORG						
50-53	Benitez et al. (2006)		CC, ORG						
48, 49	Benitez et al. (2006)		ORG						
54, 55	Benitez et al. (2006)		CC, ORG						
43	Benitez et al. (2006)	NT	NT						
44	Benitez et al. (2006)	ORG							
45	Benitez et al. (2006)	ORG							
46	Benitez et al. (2006)	ORG							
47	Benitez et al. (2006)	ORG							
48	Benitez et al. (2006)	ORG							
49	Benitez et al. (2006)	ORG							
50	Benitez et al. (2006)	CC, ORG							
51	Benitez et al. (2006)	CC, ORG							
52	Benitez et al. (2006)	CC, ORG							
53	Benitez et al. (2006)	CC, ORG							
54	Benitez et al. (2006)	CC, ORG							
55	Benitez et al. (2006)	CC, ORG							
56	Bessam and Mrabet (2003)	NT	NT	1.13	0.25	0.88	CR		
57	Campos-Herrera et al. (2010)	ORG	ORG						
58	Campos-Herrera et al. (2010)	ORG	ORG						
59	Campos-Herrera et al. (2010)	ORG	ORG						

60	Canali et al. (2003)	ORG	ORG						
61-63	Canali et al. (2004)		OA, ORG	0.95	0.00	0.95	manure and compost	1.03	1.03
61	Canali et al. (2004)	OA, ORG		0.48	0.00	0.48	compost	1.14	1.14
62	Canali et al. (2004)	OA, ORG		1.43	0.00	1.43	compost	1.00	1.00
63	Canali et al. (2004)	OA, ORG		0.93	0.00	0.93	manure	0.96	0.96
64	Canali et al. (2009)	RMP, ORG	RMP, ORG						
65, 66	Carbonell-Bojollo et al. (2009)		OA	2.12	0.00	2.12	compost OMW	6.95	6.95
65	Carbonell-Bojollo et al. (2009)	OA		2.60	0.00	2.60	compost OMW	9.30	9.30
66	Carbonell-Bojollo et al. (2009)	OA		1.63	0.00	1.63	compost OMW	4.60	4.60
67	Castro et al. (2008)	CC	CC	0.27	0.58	-0.31	CC	2.35	0.19
68	Castro et al. (2008)	CC	CC	0.69	0.58	0.11	CC	2.29	0.12
69	Castro et al. (2008)	NT	NT	0.00	0.58	-0.58	none specified	0.00	-2.17
70	Celik et al. (2004)	Control	Control	1.46	1.84	-0.38	none specified		
71	Celik et al. (2004)	OA	OA	-0.05	1.84	-1.89	compost	8.00	8.00
72	Celik et al. (2004)	OA	OA	0.51	1.84	-1.33	manure	6.63	6.63
73	Celik et al. (2004)	OA	OA	0.60	1.84	-1.24	compost	3.20	3.20
75-77	Clark et al. (1998)		ORG	0.54	0.15	0.39	manure + CR + CC		
76, 77	Clark et al. (1998)		RMP	0.54	0.08	0.46	manure + CR + CC		
74	Clark et al. (1998)	CC	CC	0.30	0.00	0.30	CR + CC		
75	Clark et al. (1998)	OA, ORG	OA	0.54	0.30	0.24	manure + CR + CC		
76	Clark et al. (1998)	RMP, ORG		0.54	0.00	0.54	manure + CR + CC		
77	Clark et al. (1998)	RMP, ORG		0.54	0.15	0.38	manure + CR + CC		
78, 79	Cookson et al. (2006)		OA, ORG				compost		
78	Cookson et al. (2006)	OA, ORG					compost		
79	Cookson et al. (2006)	OA, ORG					compost		
80	Deria et al. (2003)	Control, ORG	Control, ORG				none specified		
81	Deria et al. (2003)	Control, ORG	Control, ORG				none specified		
82	Deria et al. (2003)	Control, ORG	Control, ORG				none specified		
83	Deria et al. (2003)	Control, ORG	Control, ORG				none specified		
84	Deria et al. (2003)	OA, ORG	OA, ORG				compost		

85	Deria et al. (2003)	OA, ORG	OA, ORG				compost		
86	Deria et al. (2003)	OA, ORG	OA, ORG				compost		
87	Drinkwater et al. (1995)	RMP, ORG	RMP, ORG	0.91	0.00	0.91	manure + CR + CC		
88, 89	Efthimiadou et al. (2010)		OA, ORG	-0.83	0.00	-0.83	manure		
88	Efthimiadou et al. (2010)	OA, ORG		-1.56	0.00	-1.56	manure		
89	Efthimiadou et al. (2010)	OA, ORG		-0.09	0.00	-0.09	manure		
90-97	Fernández et al. (2009)		LT	4.25	-0.87	5.12		7.95	7.95
90	Fernández et al. (2009)	LT		5.75	-0.87	6.62	MSW/SS	3.62	3.62
91	Fernández et al. (2009)	LT		13.03	-0.87	13.90	MSW/SS	14.48	14.48
92	Fernández et al. (2009)	LT		4.75	-0.87	5.62	MSW/SS	5.92	5.92
93	Fernández et al. (2009)	LT		7.54	-0.87	8.41	MSW/SS	23.68	23.68
94	Fernández et al. (2009)	LT		-0.64	-0.87	0.23	MSW/SS	1.21	1.21
95	Fernández et al. (2009)	LT		0.06	-0.87	0.93	MSW/SS	4.83	4.83
96	Fernández et al. (2009)	LT		0.44	-0.87	1.31	MSW/SS	1.97	1.97
97	Fernández et al. (2009)	LT		3.09	-0.87	3.96	MSW/SS	7.89	7.89
98	Fernández-Ugalde et al. (2009)	NT	NT	0.84	0.00	0.84	CR		
100, 101	Gacía-Gil et al. (2000)		LT	2.57	-0.32	2.89			
99	Gacía-Gil et al. (2000)	Control	Control	-0.61	-0.32	-0.28	none specified		
100	Gacía-Gil et al. (2000)	LT		0.89	-0.32	1.21	MSW/SS		
101	Gacía-Gil et al. (2000)	LT		4.24	-0.32	4.57	MSW/SS		
102	Gacía-Gil et al. (2000)	OA	OA	0.50	-0.32	0.82	manure		
103	García-Ruiz et al. (2009)	OA, ORG	OA, ORG				manure		
104	García-Ruiz et al. (2009)	RMP, ORG	RMP, ORG				manure + CC		
105	García-Ruiz et al. (2009)	RMP, ORG	RMP, ORG				manure + CC		
106	Gómez et al. (1999)	NT	NT	0.04	0.00	0.04	none specified		
107	Gómez et al. (2009)	NT	NT	-0.15	0.22	-0.38	none specified		
108	Gómez et al. (2009)	RMP	RMP	0.56	0.22	0.34	CC		
109	Herencia et al. (2007)	RMP, ORG	RMP, ORG	2.10	-0.07	2.17	compost + CR	5.50	5.50
110	Herencia et al. (2007)	RMP, ORG	RMP, ORG	2.15	0.09	2.06	compost + CR	5.50	5.50
111	Herencia et al. (2007)	RMP, ORG	RMP, ORG	1.84	0.17	1.67	compost + CR	5.50	5.50
112	Herencia et al. (2011)	RMP, ORG	RMP, ORG	1.22	0.02	1.20	compost + CR	5.50	5.50
113	Herencia et al. (2011)	RMP, ORG	RMP, ORG	0.98	-0.51	1.48	compost + CR	5.50	5.50
114	Herencia et al. (2011)	RMP, ORG	RMP, ORG	0.70	-0.55	1.25	compost + CR	5.50	5.50

115-117	Hernández et al. (2005)		CC	0.25	-0.52	0.77	CC		
115	Hernández et al. (2005)	CC		0.40	-0.52	0.93	CC		
116	Hernández et al. (2005)	CC		0.52	-0.52	1.04	CC		
117	Hernández et al. (2005)	CC		-0.18	-0.52	0.34	CC		
118	Hernández et al. (2005)	NT	NT	-0.66	-0.52	-0.13	none specified		
119	Hernanz et al. (2009)	NT	NT	0.59	0.23	0.36	CR		
120	Hernanz et al. (2009)	RT	RT	0.23	0.23	0.00	CR		
121-125	Hojati et al. (2006)		LT	9.20	0.10	9.10	manure, SS	13.41	13.41
121	Hojati et al. (2006)	Control	Control	0.00	0.10	-0.10	none specified		
122	Hojati et al. (2006)	LT		7.73	0.10	7.63	manure	15.59	15.59
123	Hojati et al. (2006)	LT		7.50	0.10	7.40	SS	11.24	11.24
124	Hojati et al. (2006)	LT		6.09	0.10	5.99	Manure + SS	5.37	5.37
125	Hojati et al. (2006)	LT		15.50	0.10	15.40	Manure + SS	21.46	21.46
127-130	Kong et al. (2005)		CC	0.06	-0.07	0.13	CR + CC	3.05	0.95
134-136	Kong et al. (2005)		ORG	0.56	-0.01	0.57	compost + CR + CC	8.96	4.30
135, 136	Kong et al. (2005)		RMP	0.56	-0.01	0.57	compost + CR + CC	8.96	4.27
126	Kong et al. (2005)	CC	CC	0.06	-0.05	0.11	CR + CC	3.04	2.01
127	Kong et al. (2005)	CC		0.06	-0.19	0.25	CR + CC	3.05	2.14
128	Kong et al. (2005)	CC		0.06	0.04	0.02	CR + CC	3.05	1.87
129	Kong et al. (2005)	CC		0.06	-0.06	0.12	CR + CC	3.05	-1.15
130	Kong et al. (2005)	CC		-0.01	0.04	-0.05	CR + CC	4.59	-0.59
131	Kong et al. (2005)	CC	CC	-0.01	-0.06	0.05	CR + CC	4.59	0.39
132	Kong et al. (2005)	Control	Control	-0.35	-0.05	-0.30	CR	0.83	-0.20
133	Kong et al. (2005)	Control	Control	-0.19	0.04	-0.23	CR	0.91	0.08
134	Kong et al. (2005)	OA, ORG	OA	0.56	-0.01	0.57	compost + CR + CC	8.96	4.37
135	Kong et al. (2005)	RMP, ORG		0.56	0.04	0.52	compost + CR + CC	8.96	3.78
136	Kong et al. (2005)	RMP, ORG		0.56	-0.06	0.62	compost + CR + CC	8.96	4.76
137	Laudicina et al. (2010)	OA	OA	1.13	0.00	1.13	compost	4.74	2.37
138	Laudicina et al. (2010)	RMP	RMP	2.61	0.00	2.61	compost	4.74	2.37
139	Laudicina et al. (2010)	RT	RT	0.53	0.00	0.53	compost	2.37	0.00

142, 143	Lithourgidis et al. (2007)		Slurry	0.10	0.60	-0.50	Slurry		
140, 141	Lithourgidis et al. (2007)		Control	0.10	0.60	-0.50	none specified		
140	Lithourgidis et al. (2007)	Control		0.10	0.80	-0.70	none specified		
141	Lithourgidis et al. (2007)	Control		0.10	0.40	-0.30	none specified		
142	Lithourgidis et al. (2007)	Slurry		0.10	0.40	-0.30	Slurry		
143	Lithourgidis et al. (2007)	Slurry		0.10	0.80	-0.70	Slurry		
144, 145	López et al. (1996)		LT	30.40	0.00	30.40	OMW		
144	López et al. (1996)	LT		25.51	0.00	25.51	OMW		
145	López et al. (1996)	LT		35.29	0.00	35.29	OMW		
146	López-Bellido et al. (2010)	NT	NT	1.11	0.97	0.14	CR	3.52	-0.36
147	López-Bellido et al. (2010)	NT	NT	0.98	0.84	0.15	CR	2.00	0.05
148	López-Bellido et al. (2010)	NT	NT	1.27	0.52	0.76	CR	1.80	0.00
149	López-Bellido et al. (2010)	NT	NT	0.81	0.82	-0.01	CR	1.14	-0.03
150	López-Bellido et al. (2010)	NT	NT	1.35	0.31	1.05	CR	3.47	-0.24
151	López-Garrido et al. (2009)	NT	NT	0.64	-0.18	0.82	CR		
152	Lopez-Piñeiro et al. (2008)	LT	LT	10.58	-0.78	11.36	OMW	16.63	16.63
153	Lopez-Piñeiro et al. (2008)	OA	OA	3.44	-0.78	4.22	OMW	8.31	8.31
154, 155	Lopez-Piñeiro et al. (2010)		LT	9.86	-1.07	10.93	OMW	17.30	17.30
154	Lopez-Piñeiro et al. (2010)	LT		7.73	-1.07	8.79	OMW	11.53	11.53
155	Lopez-Piñeiro et al. (2010)	LT		12.00	-1.07	13.06	OMW	23.06	23.06
156-158	Madejón et al. (2003)		LT	4.17	0.38	3.79	MSW/SS	4.90	4.90
156	Madejón et al. (2003)	LT		4.52	0.38	4.14	MSW/SS	6.78	6.78
157	Madejón et al. (2003)	LT		3.82	0.38	3.44	MSW/SS	3.03	3.03
158	Madejón et al. (2003)	OA	OA	3.10	0.38	2.72	compost	2.44	2.44
159	Marinari et al. (2010a)	OA, ORG	OA, ORG	0.44	0.00	0.44	compost	2.50	2.50
160	Marinari et al. (2010a)	OA, ORG	OA, ORG	0.35	0.00	0.35	compost	2.50	2.50
161	Marinari et al. (2010b)	RMP, ORG	RMP, ORG	0.37	-0.26	0.62	compost + CR + CC	3.84	1.56
162	Martínez et al. (2008)	NT	NT	3.66	3.22	0.44	CR		
163	Martín-Rueda et al. (2007)	NT	NT	1.95	-0.36	2.31	CR		
164	Martín-Rueda et al. (2007)	RT	RT	1.64	-0.36	1.99	CR		
167, 168	Matsi et al. (2003)		Slurry	-0.28	-0.48	0.20	Slurry		
165, 166	Matsi et al. (2003)		Control	-1.83	-0.48	-1.35	none specified		
165	Matsi et al. (2003)	Control		-1.83	-1.34	-0.49	none specified		

166	Matsi et al. (2003)	Control		-1.83	0.38	-2.21	none specified		
167	Matsi et al. (2003)	Slurry		-0.28	-1.34	1.06	Slurry		
168	Matsi et al. (2003)	Slurry		-0.28	0.38	-0.66	Slurry		
169	Mazzoncini et al. (2010)	CC, ORG	CC, ORG	0.09	-0.92	1.01	CR + CC	4.23	1.87
170-172	Mazzoncini et al. (2011)		CC	0.34	0.09	0.25	CR + CC	3.44	0.36
170	Mazzoncini et al. (2011)	CC		0.17	0.09	0.08	CR + CC	3.18	0.60
171	Mazzoncini et al. (2011)	CC		0.41	0.09	0.32	CR + CC	3.48	0.30
172	Mazzoncini et al. (2011)	CC		0.43	0.09	0.34	CR + CC	3.65	0.17
173	Mazzoncini et al. (2011)	NT	NT	0.61	-0.06	0.67		2.97	2.30
174	Melero et al. (2006)	RMP, ORG	RMP, ORG	3.54	0.18	3.36	compost + CR	3.15	3.15
175, 176	Melero et al. (2007)		OA, ORG	5.46	1.45	4.01		5.17	5.17
175	Melero et al. (2007)	OA, ORG		5.18	1.45	3.73	compost	4.76	4.76
176	Melero et al. (2007)	OA, ORG		5.73	1.45	4.28	compost	5.58	5.58
177, 178	Melero et al. (2008)		OA, ORG	3.70	0.56	3.14		5.15	5.15
177	Melero et al. (2008)	OA, ORG		3.56	0.56	2.99	compost	5.04	5.04
178	Melero et al. (2008)	OA, ORG		3.84	0.56	3.28	compost	5.26	5.26
179	Melero et al. (2009)	NT	NT	0.20	0.00	0.20	CR		
180	Melero et al. (2009)	RT	RT	0.06	0.00	0.06	CR		
181-183	Monokrousos et al. (2006)		OA, ORG	-0.22	0.00	-0.22			
181	Monokrousos et al. (2006)	OA, ORG		0.44	0.00	0.44	compost		
182	Monokrousos et al. (2006)	OA, ORG		-0.34	0.00	-0.34	compost		
183	Monokrousos et al. (2006)	OA, ORG		-0.77	0.00	-0.77	compost		
184	Montanaro et al. (2009)	RMP	RMP	1.79	1.84	-0.05	compost + CC	8.33	6.31
185	Montanaro et al. (2009)	RMP	RMP	0.10	-0.72	0.82	compost + CC	9.60	8.31
187-189	Montemurro et al. (2010a)		OA	3.70	0.87	2.83	AIW, compost OMW	1.73	1.73
186	Montemurro et al. (2010a)	Control	Control	-2.25	0.87	-3.12	none specified	0.00	0.00
187	Montemurro et al. (2010a)	OA		3.12	0.87	2.25	compost	1.03	1.03
188	Montemurro et al. (2010a)	OA		0.87	0.87	0.00	AIW	2.10	2.10
189	Montemurro et al. (2010a)	OA		7.11	0.87	6.24	compost OMW	2.05	2.05
190	Montemurro et al. (2010a)	OA	OA	-1.73	0.87	-2.60	AIW	0.09	0.09
191, 192	Montemurro et al. (2010b)		LT	9.86	4.85	5.01	MSW/SS, compost OMW		
193, 194	Montemurro et al. (2010b)		LT	6.38	2.01	4.37	compost OMW	11.36	11.36
191	Montemurro et al. (2010b)	LT		9.71	4.85	4.86	MSW/SS		

192	Montemurro et al. (2010b)	LT		10.01	4.85	5.16	compost OMW		
193	Montemurro et al. (2010b)	LT		7.46	2.01	5.45	MSW/SS	4.68	4.68
194	Montemurro et al. (2010b)	LT		5.30	2.01	3.29	compost OMW	18.04	18.04
195	Moreno et al. (2006)	RT	RT	0.00	-0.02	0.02	CR		
197-201	Morra et al. (2010)		LT	1.24	-2.93	4.16		7.15	7.15
196	Morra et al. (2010)	Control	Control	-1.38	-2.93	1.55	none specified		
197	Morra et al. (2010)	LT		1.03	-2.93	3.95	compost MSW	4.50	4.50
198	Morra et al. (2010)	LT		3.30	-2.93	6.23	compost MSW	9.00	9.00
199	Morra et al. (2010)	LT		3.83	-2.93	6.75	compost MSW	13.25	13.25
200	Morra et al. (2010)	LT		-1.20	-2.93	1.73	compost MSW	4.50	4.50
201	Morra et al. (2010)	LT		-0.78	-2.93	2.15	compost MSW	4.50	4.50
202	Moussa Machraoui et al. (2010)	NT	NT	1.14	0.00	1.14	CR		
203	Moussa Machraoui et al. (2010)	NT	NT	1.11	0.00	1.11	CR		
204	Moussa Machraoui et al. (2010)	NT	NT	1.21	0.00	1.21	CR		
205	Moussa Machraoui et al. (2010)	NT	NT	0.62	0.00	0.62	CR		
206	Moussa Machraoui et al. (2010)	NT	NT	0.60	0.00	0.60	CR		
207	Mrabet et al. (2001)	NT	NT	0.41	0.10	0.31	CR		
208-210	Muñoz et al. (2007)		RMP	0.59	0.00	0.59	CR + CC		
208	Muñoz et al. (2007)	NT	NT	0.43	0.00	0.43	CR		
209	Muñoz et al. (2007)	RMP		0.73	0.00	0.73	CR + CC		
210	Muñoz et al. (2007)	RMP		0.45	0.00	0.45	CR + CC		
211	Nieto et al. (2010)	LT	LT	15.80	0.00	15.80	OMW + CR	23.90	22.90
212	Nieto et al. (2010)	LT	LT	18.93	0.00	18.93	OMW + CR	23.90	22.90
214, 215	Okur et al. (2009)		ORG	0.49	0.12	0.37	CC, manure	3.96	3.96
213-215	Okur et al. (2009)		RMP	0.54	0.12	0.42	CC, manure	5.10	5.10
213	Okur et al. (2009)	CC, ORG	CC	0.37	0.12	0.25	CC	1.68	1.68
214	Okur et al. (2009)	RMP, ORG		0.57	0.12	0.45	manure + CC	6.80	6.80
215	Okur et al. (2009)	RMP, ORG		0.52	0.12	0.40	manure + CC	3.39	3.39
216	Pardo et al. (2009)	Control	Control	-0.23	0.79	-1.02	CR + CC		
217	Pardo et al. (2009)	Control	Control	-0.77	-1.31	0.54	CR + CC		
218	Pardo et al. (2009)	OA, ORG	OA, ORG	0.54	0.79	-0.25	compost + CR + CC	0.77	0.77
219	Pardo et al. (2009)	OA, ORG	OA, ORG	-0.65	-1.31	0.66	compost + CR + CC	0.77	0.77

220-224	Plaza et al. (2004)		Slurry	-0.07	-0.04	-0.03	Slurry	0.73	0.73
220	Plaza et al. (2004)	Control	Control	0.00	-0.04	0.04	none specified	0.00	
221	Plaza et al. (2004)	Slurry		0.04	-0.04	0.08	Slurry	0.24	0.24
222	Plaza et al. (2004)	Slurry		0.09	-0.04	0.13	Slurry	0.49	0.49
223	Plaza et al. (2004)	Slurry		0.00	-0.04	0.04	Slurry	0.73	0.73
224	Plaza et al. (2004)	Slurry		-0.09	-0.04	-0.05	Slurry	0.97	0.97
225	Plaza et al. (2004)	Slurry		-0.39	-0.04	-0.35	Slurry	1.22	1.22
226, 227	Raviv et al. (2006)		ORG	2.06	0.00	2.06	Compost, CC		
226	Raviv et al. (2006)	OA, ORG	OA	1.06	0.00	1.06	Compost		
227	Raviv et al. (2006)	RMP, ORG	RMP	3.06	0.00	3.06	Compost + CC		
228	Reganold et al. (2010)	RMP, ORG	RMP, ORG				Compost + CC		
229	Romanya and Rovira (2009)	OA, ORG	OA, ORG	-0.19	0.00	-0.19	manure		
230	Romanya and Rovira (2009)	OA, ORG	OA, ORG	0.46	0.00	0.46	manure		
231-237	Ryan et al. (2008)		RT	0.13	0.00	0.13	CR		
231	Ryan et al. (2008)	RT		0.02	0.00	0.02	CR		
232	Ryan et al. (2008)	RT		0.06	0.00	0.06	CR		
233	Ryan et al. (2008)	RT		0.14	0.00	0.14	CR		
234	Ryan et al. (2008)	RT		0.06	0.00	0.06	CR		
235	Ryan et al. (2008)	RT		0.16	0.00	0.16	CR		
236	Ryan et al. (2008)	RT		0.17	0.00	0.17	CR		
237	Ryan et al. (2008)	RT		0.28	0.00	0.28	CR		
238	Sombrero et al. (2010)	NT	NT	1.77	0.46	1.32	CR	2.04	2.04
239	Sombrero et al. (2010)	RT	RT	1.31	0.46	0.85	CR	2.01	2.01
240-243	Sommer et al. (2011)		OA	1.04	0.00	1.04			
244-251	Sommer et al. (2011)		RMP	1.60	0.00	1.60			
240	Sommer et al. (2011)	OA		2.03	0.00	2.03	compost + CR		
241	Sommer et al. (2011)	OA		0.78	0.00	0.78	compost + CR		
242	Sommer et al. (2011)	OA		1.68	0.00	1.68	compost + CR		
243	Sommer et al. (2011)	OA		-0.31	0.00	-0.31	compost + CR		
244	Sommer et al. (2011)	RMP		2.73	0.00	2.73	compost + CR		
245	Sommer et al. (2011)	RMP		1.48	0.00	1.48	compost + CR		
246	Sommer et al. (2011)	RMP		2.84	0.00	2.84	compost + CR		
247	Sommer et al. (2011)	RMP		1.59	0.00	1.59	compost + CR		

248	Sommer et al. (2011)	RMP		1.98	0.00	1.98	compost + CR		
249	Sommer et al. (2011)	RMP		-0.01	0.00	-0.01	compost + CR		
250	Sommer et al. (2011)	RMP		2.09	0.00	2.09	compost + CR		
251	Sommer et al. (2011)	RMP		0.10	0.00	0.10	compost + CR		
252	Sommer et al. (2011)	RT	RT	0.73	0.00	0.73			
253	Sommer et al. (2011)	RT	RT	0.59	0.00	0.59			
254	Sommer et al. (2011)	RT	RT	0.40	0.00	0.40	CR		
255-260	Tejada et al. (2006)		OA	6.64	-0.62	7.26			
255	Tejada et al. (2006)	OA		4.08	-0.62	4.70	compost		
256	Tejada et al. (2006)	OA		5.23	-0.62	5.85	compost		
257	Tejada et al. (2006)	OA		6.66	-0.62	7.28	compost		
258	Tejada et al. (2006)	OA		5.94	-0.62	6.57	AIW		
259	Tejada et al. (2006)	OA		7.60	-0.62	8.22	AIW		
260	Tejada et al. (2006)	OA		10.31	-0.62	10.93	AIW		
261	Vavoulidou et al. (2006)	ORG	ORG						
262	Vavoulidou et al. (2009)	ORG	ORG						
263	Vavoulidou et al. (2009)	ORG	ORG						
264	Vavoulidou et al. (2009)	ORG	ORG						
265	Vavoulidou et al. (2009)	ORG	ORG						
266	Vavoulidou et al. (2009)	ORG	ORG						
267, 268	Veenstra et al. (2007)		CC	0.83	-0.03	0.86	CC	2.09	2.09
267	Veenstra et al. (2007)	CC		0.77	-0.03	0.80	CC	1.83	1.83
268	Veenstra et al. (2007)	CC		0.90	-0.03	0.93	CC	2.34	2.34
269	Veenstra et al. (2007)	RT	RT	-0.07	-0.03	-0.04	none specified	0.00	0.00
270, 271	Virto et al. (2007)		NT	2.53	2.09	0.44			
270	Virto et al. (2007)	NT		2.55	2.09	0.46	CR		
271	Virto et al. (2007)	NT		2.51	2.09	0.42	none specified		

Appendix A3. Supplementary data of Study 3.

Table A3.1 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) cereals in Spain. Data refer to 1 ha and year unless otherwise stated

	Barley 1		Barley 2		Barley 3		Barley 4		Wheat 1		Wheat 2		Wheat 3		Oats	
	Con	Org	Con	Org	Con	Org	Con	Or	Con	Org	Con	Org	Con	Org	Con	Org
Inputs																
Drip (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %
Water (m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Electricity (KWh)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Seeds (kg	250	100	230	210	180	200	200	200	180	200	125	150	180	200	120	120
Seedlings (units)	-45	57	-44	58	-43	59	-42	60	-41	61	-40	62	-39	63	-38	64
Machinery use (hours)	10	8	8	7	7	7	8	6	7	5	9	7	10	5	6	4
Fuel consumption (liters)	171	139	132	116	109	121	136	93	103	96	141	128	164	93	122	85
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	0	0	76	0	82	0	92	0	127	0	49	0	163	0	0	0
Mineral phosphorus (kg P2O5)	0	0	96	0	60	0	0	0	60	0	30	0	60	0	0	0
Mineral potassium (kg K2O)	0	0	104	0	60	0	0	0	60	0	45	0	0	0	0	0
Manure (Mg)	0.00	0.38	0.00	3.33	0.00	3.75	0.00	2.50	0.00	0.00	0.00	1.50	0.00	0.00	0.00	0.00
Slurry (Mg)	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	228	811	0	475	0	464	0	358	0	149	0	296	0	1120	637	944
Total nitrogen inputs (kg)	105	12	76	23	82	23	92	28	127	24	49	11	163	10	6	8
Synthetic pesticides (kg active matter)	0.0	0.0	0.5	0.0	0.5	0.0	0.5	0.0	0.9	0.0	0.0	0.0	1.2	0.0	0.0	0.0
Sulphur (kg)	0	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0
Copper (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Natural pesticides (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Steel (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Greenhouse cover plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Concrete (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Production																
Yield (Mg)	1.80	1.50	2.90	2.50	3.50	2.60	4.90	2.80	2.00	1.60	1.83	1.13	5.50	2.00	1.20	1.50
Weeds (Mg)	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67
Residue yield (Mg)	1.85	1.54	2.98	2.57	3.60	2.67	5.04	2.88	2.38	1.90	2.17	1.34	6.53	2.38	1.56	1.95
Residue destiny																
Open burning (Mg)	0.02	0.00	0.04	0.03	0.04	0.03	0.06	0.03	0.03	0.02	0.03	0.02	0.08	0.00	0.00	0.00
Soil incorporation (Mg)	0.00	1.54	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.38	1.56	1.95
Coproduct (Mg)	1.83	0.00	2.95	2.54	3.56	2.64	4.98	2.84	2.35	1.88	2.14	1.32	6.45	0.00	0.00	0.00
Allocation to coproduct (%)	15%	0%	15%	15%	15%	15%	15%	15%	15%	15%	15%	15%	15%	0%	0%	0%
Direct nitrous oxide (kg N2O)	0.13	0.02	0.09	0.03	0.10	0.03	0.12	0.03	0.16	0.03	0.06	0.01	0.20	0.01	0.01	0.01
Indirect nitrous oxide (kg N2O)	0.83	0.10	0.47	0.18	0.52	0.18	0.58	0.22	0.80	0.19	0.30	0.08	1.02	0.08	0.05	0.06
Methane (kg CH4)	0.06	0.00	0.10	0.08	0.12	0.09	0.16	0.09	0.08	0.06	0.07	0.04	0.21	0.00	0.00	0.00
Carbon (kg C)	-69	-143	0	-125	0	-121	0	-89	-301	456	0	-70	0	-191	-109	-161

Table A3.2 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) legumes in Spain. Data refer to 1 ha and year unless otherwise stated

	Peas 1		Peas 2		Peas 3		Peas 4		Fava bean		Vetch	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs												
Drip (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Water (m3)	0	0	0	0	0	0	0	0	0	0	0	0
Electricity (KWh)	0	0	0	0	0	0	0	0	0	0	0	0
Seeds (kg	210	210	180	110	150	300	230	200	200	200	120	130
Seedlings (units)	-37	65	-36	66	-35	67	-34	68	-33	69	-32	70

Machinery use (hours)	3	5	5	3	5	6	8	5	10	5	5	6
Fuel consumption (liters)	67	88	84	61	90	120	139	85	163	102	102	103
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	0	0	51	0	0	0	0	0	26	0	0	0
Mineral phosphorus (kg P2O5)	0	0	35	0	0	0	90	0	0	0	0	0
Mineral potassium (kg K2O)	0	0	53	0	0	0	0	0	88	0	0	0
Manure (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	3.33
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	0	149	0	149	0	149	0	247	1979	1562	462	937
Total nitrogen inputs (kg)	35	35	86	35	35	35	35	42	128	83	46	69
Synthetic pesticides (kg active matter)	0.0	0.0	0.0	0.0	1.3	0.0	1.6	0.0	2.5	0.0	0.0	0.0
Sulphur (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Copper (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Steel (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Greenhouse cover plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Concrete (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Production												
Yield (Mg)	2.00	1.20	1.78	1.00	2.50	1.55	3.13	0.64	3.50	2.50	1.00	1.00
Weeds (Mg)	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67	0.39	0.67
Residue yield (Mg)	2.85	1.71	2.53	1.42	3.56	2.21	4.46	0.91	4.84	3.46	1.13	1.13
Residue destiny												
Open burning (Mg)	0.57	0.34	0.51	0.28	0.71	0.44	0.89	0.18	0.00	0.00	0.00	0.00
Soil incorporation (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.84	3.46	1.13	1.13
Coproduct (Mg)	2.28	1.37	2.02	1.14	2.85	1.77	3.57	0.73	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	11%	11%	11%	11%	11%	11%	11%	11%	0%	0%	0%	0%
Direct nitrous oxide (kg N2O)	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.01	0.12	0.06	0.01	0.04

Indirect nitrous oxide (kg N2O)	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.06	0.69	0.38	0.09	0.27
Methane (kg CH4)	1.54	0.92	1.36	0.77	1.92	1.19	2.41	0.49	0.00	0.00	0.00	0.00
Carbon (kg C)	0	-25	0	-25	0	-25	0	-55	-338	-267	-79	-204

Table A3.3 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) rice in Spain. Data refer to 1 ha and year unless otherwise stated

	Rice 1		Rice 2		Rice 3	
	Con	Org	Con	Org	Con	Org
Inputs						
Drip (%)	0%	0%	0%	0%	0%	0%
Surface (%)	100%	100%	100%	100%	100%	100%
Rainfed (%)	0%	0%	0%	0%	0%	0%
Water (m3)	11500	11500	11500	11500	11500	11500
Electricity (KWh)	3634	3634	3634	3634	3634	3634
Seeds (kg	200	230	180	175	264	300
Seedlings (units)	-31	71	-30	72	-29	73
Machinery use (hours)	12	12	13	9	10	10
Fuel consumption (liters)	212	226	227	167	194	188
Mulching plastic (kg)	0	0	0	0	0	0
Mineral nitrogen (kg N)	121	0	244	0	92	0
Mineral phosphorus (kg P2O5)	23	0	34	0	0	0
Mineral potassium (kg K2O)	23	0	34	0	0	0
Manure (Mg)	0.00	9.00	0.00	9.00	0.00	5.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	3126	2941	0	3294	3573	3278
Total nitrogen inputs (kg)	156	122	244	88	132	60
Synthetic pesticides (kg active matter)	6.5	0.0	7.0	0.0	7.9	0.0
Sulphur (kg)	0	0	0	0	0	0

Copper (kg)	0	0	1	0	0	0
Natural pesticides (kg)	0	1	0	0	0	0
Steel (kg)	0	0	0	0	0	0
Greenhouse cover plastic (kg)	0	0	0	0	0	0
Concrete (kg)	0	0	0	0	0	0
Production						
Yield (Mg)	7.00	4.50	7.50	5.00	8.00	6.00
Weeds (Mg)	0.30	0.64	0.30	0.64	0.30	0.64
Residue yield (Mg)	7.64	4.91	8.19	5.46	8.74	6.55
Residue destiny						
Open burning (Mg)	0.00	0.00	8.19	0.00	0.00	0.00
Soil incorporation (Mg)	7.64	4.91	0.00	5.46	8.74	6.55
Coproduct (Mg)	0.00	0.00	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	0%	0%	0%	0%	0%	0%
Direct nitrous oxide (kg N2O)	0.73	0.57	1.15	0.42	0.62	0.28
Indirect nitrous oxide (kg N2O)	1.03	0.95	1.53	0.69	0.89	0.47
Methane (kg CH4)	398.88	438.97	222.31	449.71	421.60	433.12
Carbon (kg C)	-534	-603	0	-680	-610	-616

Table A3.4 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) open-air vegetables in Spain. Data refer to 1 ha and year unless otherwise stated

	Asparragus		Lettuce 1		Lettuce 2		Melon 1		Melon 2		Celery	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs												
Drip (%)	0%	0%	100%	100%	0%	0%	100%	100%	0%	0%	100%	100%
Surface (%)	100%	100%	0%	0%	100%	100%	0%	0%	0%	0%	0%	0%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	100%	100%	0%	0%
Water (m3)	10000	10000	1200	1200	900	900	4800	4800	0	0	5000	5000
Electricity (KWh)	3160	3160	379	379	284	284	1517	1517	0	0	1580	1580
Seeds (kg	0	0	90000	###	###	50000	5000	5000	0	0	80000	###

Seedlings (units)	-28	74	-27	75	-26	76	-25	77	-24	78	-23	79
Machinery use (hours)	25	26	18	27	25	14	14	5	17	18	20	20
Fuel consumption (liters)	58	209	259	118	329	221	180	52	244	305	229	250
Mulching plastic (kg)	0	0	0	0	138	0	138	138	40	0	0	0
Mineral nitrogen (kg N)	223	0	82	0	3	0	54	0	28	0	78	0
Mineral phosphorus (kg P2O5)	180	0	40	0	19	0	46	0	42	0	42	0
Mineral potassium (kg K2O)	210	38	136	0	15	0	67	0	84	0	77	0
Manure (Mg)	0.00	10.00	10.00	15.00	10.00	10.00	0.00	2.00	0.00	0.00	0.00	15.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	78.33
Total carbon inputs (kg)	0	4347	836	2301	840	2146	195	1215	0	1047	0	2525
Total nitrogen inputs (kg)	223	262	192	165	63	80	72	12	28	24	78	107
Synthetic pesticides (kg active matter)	5.7	0.0	0.0	0.0	7.7	0.0	12.6	0.0	0.0	0.0	8.2	0.0
Sulphur (kg)	0	0	0	0	0	0	30	0	0	0	0	0
Copper (kg)	1	1	0	0	0	0	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	0	0	0	0	0	0	0	0	10
Steel (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Greenhouse cover plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Concrete (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Production												
Yield (Mg)	8.00	6.80	32.00	29.00	32.00	11.00	10.00	8.00	6.00	5.00	37.00	30.00
Weeds (Mg)	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00
Residue yield (Mg)	8.62	7.32	1.52	1.38	1.52	0.52	0.66	0.53	0.40	0.33	0.59	0.48
Residue destiny												
Open burning (Mg)	8.62	0.00	0.30	0.28	0.30	0.00	0.00	0.00	0.08	0.07	0.12	0.10
Soil incorporation (Mg)	0.00	7.32	0.00	0.00	0.00	0.52	0.66	0.00	0.00	0.00	0.00	0.00
Coproduct (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Direct nitrous oxide (kg	1.05	1.23	1.99	1.71	1.00	1.27	0.75	0.12	0.04	0.03	0.81	1.11

N2O)												
Indirect nitrous oxide (kg N2O)	1.40	2.06	1.38	1.30	0.49	0.63	0.48	0.09	0.18	0.19	0.49	0.84
Methane (kg CH4)	23.26	0.00	0.82	0.75	0.82	0.00	0.00	0.00	0.21	0.18	0.32	0.26
Carbon (kg C)	0	-873	-255	-561	-256	-493	-33	-230	0	303	0	-629

Table A3.4 (Continued) Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) open-air vegetables in Spain. Data refer to 1 ha and year unless otherwise stated

	Coliflower		Potato		Broccoli		Onion 1		Onion 2		Beans 1		Beans 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs														
Drip (%)	100%	100%	0%	0%	100%	100%	0%	0%	0%	0%	100%	100%	100%	100%
Surface (%)	0%	0%	100%	100%	0%	0%	100%	100%	100%	100%	0%	0%	0%	0%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Water (m3)	1000	1000	10000	10000	900	900	4300	4300	4300	4300	2700	2700	2700	2700
Electricity (KWh)	316	316	3160	3160	284	284	1359	1359	1359	1359	853	853	853	853
Seeds (kg	16700	25000	1380	960	50000	50000	90000	###	###	95000	45	40	25	40
Seedlings (units)	-22	80	-21	81	-20	82	-19	83	-18	84	-17	85	-16	86
Machinery use (hours)	36	19	42	13	19	19	37	25	40	16	28	27	27	32
Fuel consumption (liters)	545	273	251	183	230	278	404	390	437	208	182	147	149	214
Mulching plastic (kg)	0	0	0	0	138	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	141	0	135	0	344	0	188	0	73	0	7	0	0	0
Mineral phosphorus (kg P2O5)	38	0	45	0	127	0	75	0	30	0	15	0	0	0
Mineral potassium (kg K2O)	101	0	270	0	483	0	215	0	120	0	50	0	0	0
Manure (Mg)	10.50	25.00	10.50	48.00	0.00	15.00	0.00	18.00	15.00	50.00	0.00	12.00	10.00	15.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	13.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	1573	3860	882	5751	1	2517	2198	4960	1470	5947	0	2223	980	2517
Total nitrogen inputs (kg)	250	199	198	336	344	105	335	270	178	350	42	119	105	140
Synthetic pesticides (kg active matter)	4.8	0.0	1.5	0.0	6.1	0.0	14.8	0.0	7.9	0.0	0.9	0.0	0.7	0.0

Sulphur (kg)	0	0	0	0	0	0	0	0	0	0	0	3	0	0
Copper (kg)	35	0	0	0	0	0	0	0	3	2	0	0	0	0
Natural pesticides (kg)	0	21	0	1	0	19	0	0	0	0	0	0	0	0
Steel (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Greenhouse cover plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Concrete (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Production														
Yield (Mg)	15.00	10.00	30.00	24.00	20.00	11.00	50.00	50.00	50.00	35.00	12.00	6.00	1.75	1.50
Weeds (Mg)	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00
Residue yield (Mg)	1.69	1.12	4.20	3.36	1.93	1.06	6.14	6.14	6.14	4.30	5.76	2.88	0.84	0.72
Residue destiny														
Open burning (Mg)	0.00	0.00	2.10	1.68	0.39	0.21	0.00	0.00	1.23	0.86	1.15	0.58	0.17	0.14
Soil incorporation (Mg)	1.69	1.12	0.00	0.00	0.00	0.00	6.14	6.14	0.00	0.00	0.00	0.00	0.00	0.00
Coproduct (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Direct nitrous oxide (kg N2O)	2.59	2.06	3.14	5.33	3.57	1.09	5.32	4.28	2.82	5.56	0.08	0.87	0.73	1.09
Indirect nitrous oxide (kg N2O)	1.74	1.56	1.34	2.64	2.16	0.83	2.34	2.12	1.28	2.75	0.05	0.66	0.55	0.83
Methane (kg CH4)	0.00	0.00	5.67	4.54	1.04	0.57	0.00	0.00	3.32	2.32	3.11	1.56	0.45	0.39
Carbon (kg C)	-406	-987	-269	-1611	0	-626	-375	-1076	-448	-1671	0	-537	-298	-626

Table A3.5 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) greenhouse vegetables in Spain. Data refer to 1 ha and year unless otherwise stated

	Cherry tomato GH		Tomato GH 1		Tomato GH 2		Tomato GH 3		Tomato GH 4		Lettuce GH		Pepper GH		Beans GH	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs																
Drip (%)	100%	100%	100%	100%	100%	100%	100%	100%	100 %	100%	100 %	100 %	100%	100%	100 %	100 %
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Water (m3)	4000	4000	3700	3700	3700	3700	3700	3700	5300	5300	900	900	6200	6200	1400	1400

Electricity (KWh)	1264	1264	1169	1169	1169	1169	1169	1169	1675	1675	284	284	1959	1959	442	442
	3300	3300	1900	1900	3000	3000	2000	2000					2500	2500		
Seeds (kg	0	0	0	0	0	0	0	0	8190	10700	###	###	0	0	30	30
Seedlings (units)	-15	87	-14	88	-13	89	-12	90	-11	91	-10	92	-9	93	-8	94
Machinery use (hours)	135	133	10	5	12	17	18	14	51	61	16	31	26	15	8	11
Fuel consumption (liters)	978	991	166	87	163	97	123	73	388	450	125	196	342	210	102	145
Mulching plastic (kg)	0	0	80	218	218	218	218	218	138	138	138	0	471	471	218	80
Mineral nitrogen (kg N)	189	0	304	0	174	0	0	0	1147	0	29	0	191	0	77	0
Mineral phosphorus (kg P2O5)	196	0	59	0	120	0	0	0	786	0	45	0	224	0	94	0
Mineral potassium (kg K2O)	194	0	440	0	645	0	0	0	296	0	57	0	612	0	94	0
Manure (Mg)	12.00	35.00	0.00	0.00	5.00	15.00	15.00	30.00	1.50	5.00	0.00	2.50	15.00	15.00	7.50	7.50
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	###	0.00	###	0.00	0.00	###	0.00	0.00	###	0.00	###	0.00	72.00	0.00	0.00
Total carbon inputs (kg)	3453	6408	0	1753	418	2517	1309	3567	147	1548	0	1784	6307	2524	735	1782
Total nitrogen inputs (kg)	462	412	304	216	229	105	105	180	1157	38	29	92	696	107	165	88
Synthetic pesticides (kg active matter)	1.4	0.0	2.2	0.0	11.3	0.0	0.4	0.0	394.3	0.0	3.2	0.0	4.9	0.0	1.2	0.0
Sulphur (kg)	0	259	0	0	0	0	6	3	18	71	0	3	0	3	0	3
Copper (kg)	0	1	4	0	0	0	0	7	23	9	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	38	0	5	0	20	0	133	0	20	0	19	0	0
Steel (kg)	411	411	411	411	411	411	411	411	411	411	411	411	411	411	411	411
Greenhouse cover plastic (kg)	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208	1208
Concrete (kg)	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290	7290
Production																
Yield (Mg)	57.60	48.00	95.00	70.00	80.00	37.50	###	55.00	80.22	63.16	30.00	30.00	70.00	60.00	17.00	20.00
Weeds (Mg)	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00	0.00	2.00
Residue yield (Mg)	7.19	5.99	11.86	8.74	9.98	4.68	12.48	6.86	10.01	7.88	1.43	1.43	15.27	13.09	8.16	9.60
Residue destiny																
Open burning (Mg)	0.00	0.00	2.37	1.75	2.00	0.94	2.50	1.37	2.00	1.58	0.29	0.00	0.00	2.62	1.63	1.92
Soil incorporation (Mg)	7.19	5.99	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.43	15.27	0.00	0.00	0.00

Coproduct (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Direct nitrous oxide (kg N2O)	4.79	4.27	3.15	2.24	2.38	1.09	1.09	1.87	12.00	0.40	0.30	0.95	7.22	1.11	1.34	0.54
Indirect nitrous oxide (kg N2O)	3.33	3.24	1.91	1.70	1.53	0.83	0.83	1.41	7.29	0.30	0.18	0.72	5.17	0.84	0.90	0.41
Methane (kg CH4)	0.00	0.00	6.40	4.72	5.39	2.53	6.74	3.71	5.41	4.26	0.77	0.00	0.00	7.07	4.41	5.18
Carbon (kg C)	-747	-1557	0	-394	-127	-626	-399	-946	-45	-331	0	-347	-1273	-629	-224	-403

Table A3.6 Global warming potential of organic and conventional Spanish cereals for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Barley 1		Barley 2		Barley 3		Barley 4		Wheat 1		Wheat 2		Wheat 3		Oats	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2-eq/ha																
Machinery production	38	24	27	25	20	29	31	17	21	19	27	27	35	20	24	15
Fuel production	74	60	57	50	47	52	59	40	45	42	61	55	71	40	53	37
Fuel use	462	376	357	313	294	326	366	250	279	260	380	344	442	252	330	228
Fertilizer production	94	1	499	13	484	14	398	9	702	0	281	6	825	0	0	0
Direct nitrous oxide	39	5	28	9	31	8	34	10	48	9	18	4	61	4	2	3
Indirect nitrous oxide	246	29	142	55	154	53	172	64	238	56	91	25	304	24	15	19
Pesticides	0	0	2	0	1	0	1	0	7	0	0	0	10	0	0	0
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	1	0	2	2	3	2	4	2	2	2	2	1	5	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	88	23	81	48	64	45	71	45	64	64	44	48	64	64	42	31
Carbon	-127	-261	0	-229	0	-222	0	-163	-552	836	0	-129	0	-350	-199	-295
Total	916	256	1196	285	1099	308	1137	277	853	1288	905	382	1818	53	268	38

Product-based emissions (kg CO2-eq-ha)																
Machinery production	18	16	8	8	5	10	5	5	9	10	13	21	5	10	20	10
Fuel production	35	40	17	17	12	17	10	12	19	22	28	42	11	20	44	24
Fuel use	219	250	105	107	72	107	64	76	119	138	177	261	68	126	275	152
Fertilizer production	45	1	147	4	118	5	69	3	299	0	131	4	128	0	0	0
Direct nitrous oxide	19	3	8	3	8	3	6	3	20	5	8	3	9	2	2	2
Indirect nitrous oxide	117	19	42	19	38	17	30	20	101	30	42	19	47	12	13	13
Pesticides	0	0	0	0	0	0	0	0	3	0	0	0	1	0	0	0
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	1	0	1	1	1	1	1	1	1	1	1	1	1	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	42	15	24	16	16	15	12	14	27	34	21	36	10	32	35	21
Carbon	-60	-174	0	-78	0	-73	0	-50	-235	445	0	-97	0	-175	-166	-197
Total	435	171	352	97	268	101	198	85	363	685	422	289	281	27	224	25
Coproduct	74	0	60	17	46	17	34	14	64	120	74	51	49	0	0	0

Table A3.7 Global warming potential of organic and conventional Spanish legumes for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Peas 1		Peas 2		Peas 3		Peas 4		Fava bean		Vetch	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e/ha)												
Machinery production	11	17	14	10	16	28	30	15	35	24	20	21
Fuel production	29	38	36	27	39	52	60	37	70	44	44	44
Fuel use	181	238	225	166	244	322	376	228	438	275	275	277
Fertilizer production	0	0	302	0	0	0	368	4	429	0	0	13
Direct nitrous oxide	0	0	19	0	0	0	0	3	35	18	4	13
Indirect nitrous oxide	0	0	96	0	0	0	0	16	205	112	25	80

Pesticides	0	0	0	0	15	0	17	0	24	0	0	0
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0
Methane	38	23	34	19	48	30	60	12	0	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	49	49	42	25	35	69	53	46	46	43	28	28
Carbon	0	-46	0	-46	0	-46	0	-101	-619	-489	-145	-373
Total	321	325	778	206	412	464	983	264	663	28	252	103
Product-based emissions (g CO_2e/kg)												
Machinery production	5	13	7	9	6	16	9	21	10	10	20	21
Fuel production	13	28	18	24	14	30	17	51	20	18	44	44
Fuel use	80	175	112	147	86	184	106	316	125	110	275	277
Fertilizer production	0	0	150	0	0	0	104	5	122	0	0	13
Direct nitrous oxide	0	0	10	0	0	0	0	4	10	7	4	13
Indirect nitrous oxide	0	0	48	0	0	0	0	23	59	45	25	80
Pesticides	0	0	0	0	5	0	5	0	7	0	0	0
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0
Methane	17	17	17	17	17	17	17	17	0	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	22	36	21	23	12	40	15	64	13	17	28	28
Carbon	0	-34	0	-41	0	-27	0	-140	-177	-195	-145	-373
Total	142	240	388	183	146	265	278	366	190	11	252	103
Coproduct	18	31	50	24	19	34	36	47	0	0	0	0

Table A3.8 Global warming potential of organic and conventional Spanish rice for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Rice 1		Rice 2		Rice 3	
	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e/ha)						
Machinery production	43	46	53	41	40	38
Fuel production	92	98	98	72	84	82
Fuel use	572	610	611	450	522	508
Fertilizer production	551	34	616	34	398	19
Direct nitrous oxide	219	171	343	124	186	84
Indirect nitrous oxide	308	285	457	206	266	141
Pesticides	70	5	81	0	93	3
Irrigation infraestructure	90	90	90	90	90	90
Irrigation energy	2018	2018	2018	2018	2018	2018
Methane	9972	10974	5558	11243	10540	10828
Greenhouse	0	0	0	0	0	0
Nursery	320	490	288	373	422	640
Carbon	-978	-1105	0	-1247	-1118	-1129
Total	13278	13716	10385	13405	13540	13322
Product-based emissions (g CO_2e/kg)						
Machinery production	6	10	7	8	5	6
Fuel production	13	22	13	14	10	14
Fuel use	82	136	82	90	65	85
Fertilizer production	79	8	82	7	50	3
Direct nitrous oxide	31	38	46	25	23	14
Indirect nitrous oxide	44	63	61	41	33	23
Pesticides	10	1	11	0	12	1
Irrigation infraestructure	13	20	12	18	11	15
Irrigation energy	288	449	269	404	252	336
Methane	1425	2439	741	2249	1318	1805
Greenhouse	0	0	0	0	0	0
Nursery	46	109	38	75	53	107

Carbon	-140	-246	0	-249	-140	-188
Total	1897	3048	1385	2681	1693	2220
Coproduct	0	0	0	0	0	0

Table A3.9 Global warming potential of organic and conventional Spanish open-air vegetables for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Asparragus		Lettuce 1		Lettuce 2		Melon 1		Melon 2		Celery	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e/ha)												
Machinery production	5	40	49	16	65	46	36	10	47	62	54	60
Fuel production	25	90	112	51	142	96	78	23	105	132	99	108
Fuel use	156	562	698	319	886	596	486	141	657	823	617	675
Fertilizer production	1284	102	887	56	111	38	411	8	220	0	565	65
Direct nitrous oxide	313	368	592	510	299	379	224	37	10	9	241	332
Indirect nitrous oxide	418	613	410	386	147	187	144	28	52	56	146	251
Pesticides	54	2	100	0	59	0	81	0	0	0	59	31
Irrigation infraestructure	34	34	311	311	144	144	311	311	0	0	311	311
Irrigation energy	1755	1755	211	211	158	158	842	842	0	0	878	878
Methane	582	0	21	19	21	0	0	0	5	4	8	6
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	0	0	363	403	403	202	40	40	0	0	323	403
Carbon	0	-1601	-467	-1028	-469	-904	-61	-974	0	555	0	-1153
Total	4806	1967	3294	1260	2260	941	2882	755	1182	1644	3304	1971
Product-based emissions (g CO_2e/kg)												
Machinery production	1	6	2	1	2	4	4	1	8	12	1	2
Fuel production	3	13	4	2	4	9	8	3	18	26	3	4
Fuel use	20	83	22	11	28	54	49	18	109	165	17	23
Fertilizer production	160	15	28	2	3	3	41	1	37	0	15	2

Direct nitrous oxide	39	54	18	18	9	34	22	5	2	2	7	11
Indirect nitrous oxide	52	90	13	13	5	17	14	4	9	11	4	8
Pesticides	7	0	3	0	2	0	8	0	0	0	2	1
Irrigation infraestructure	4	5	10	11	4	13	31	39	0	0	8	10
Irrigation energy	219	258	7	7	5	14	84	105	0	0	24	29
Methane	73	0	1	1	1	0	0	0	1	1	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	0	0	11	14	13	18	4	5	0	0	9	13
Carbon	0	-235	-15	-35	-15	-82	-6	-122	0	111	0	-38
Total	601	289	103	43	71	86	288	94	197	329	89	66
Coproduct	0	0	0	0	0	0	0	0	0	0	0	0

Table A3.9 (Continued) Global warming potential of organic and conventional Spanish open-air vegetables for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Coliflower		Potato		Broccoli		Onion 1		Onion 2		Beans 1		Beans 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e/ha)														
Machinery production	102	54	40	36	50	60	82	73	93	42	29	22	26	34
Fuel production	236	118	109	79	100	120	175	169	189	90	79	64	64	93
Fuel use	1471	737	677	494	620	749	1088	1050	1179	560	491	397	400	578
Fertilizer production	834	94	1008	180	3017	56	1208	68	511	188	198	45	38	56
Direct nitrous oxide	773	614	936	1589	1063	325	1585	1276	840	1655	23	260	216	325
Indirect nitrous oxide	519	465	400	787	645	246	697	632	382	820	14	197	164	246
Pesticides	108	73	13	8	48	34	151	0	67	3	11	1	8	0
Irrigation infraestructure	311	311	34	34	311	311	144	144	144	144	311	311	311	311
Irrigation energy	176	176	1755	1755	158	158	755	755	755	755	474	474	474	474
Methane	0	0	142	113	26	14	0	0	83	58	78	39	11	10
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Nursery	67	101	216	128	202	202	109	125	145	115	18	15	10	15
Carbon	-745	-1809	-492	-2954	-1	-1148	-688	-1973	-821	-3064	0	-984	-547	-1148
Total	3852	933	4882	2286	6535	1130	5305	2317	3591	1382	1751	852	1180	996
Product-based emissions (g CO_2e/kg)														
Machinery production	7	5	1	2	3	5	2	1	2	1	2	4	15	23
Fuel production	16	12	4	3	5	11	3	3	4	3	7	11	37	62
Fuel use	98	74	23	21	31	68	22	21	24	16	41	66	229	385
Fertilizer production	56	9	34	8	151	5	24	1	10	5	17	8	21	38
Direct nitrous oxide	52	61	31	66	53	30	32	26	17	47	2	43	124	216
Indirect nitrous oxide	35	46	13	33	32	22	14	13	8	23	1	33	94	164
Pesticides	7	7	0	0	2	3	3	0	1	0	1	0	5	0
Irrigation infraestructure	21	31	1	1	16	28	3	3	3	4	26	52	178	208
Irrigation energy	12	18	59	73	8	14	15	15	15	22	39	79	271	316
Methane	0	0	5	5	1	1	0	0	2	2	6	6	6	6
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	4	10	7	5	10	18	2	2	3	3	2	3	6	10
Carbon	-50	-181	-16	-123	0	-104	-14	-39	-16	-88	0	-164	-313	-766
Total	257	93	163	95	327	103	106	46	72	39	146	142	674	664
Coproduct	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A3.10 Global warming potential of organic and conventional Spanish greenhouse vegetables for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Cherry tomato GH		Tomato GH 1		Tomato GH 2		Tomato GH 3		Tomato GH 4		Lettuce GH		Pepper GH		Beans GH	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO_2e/ha)																
Machinery production	242	194	32	18	32	15	19	10	92	107	25	34	54	42	18	26
Fuel production	424	430	72	38	70	42	53	32	168	195	54	85	148	91	44	63

Fuel use	2636	2672	447	235	439	261	332	197	1046	1214	337	529	922	566	274	390
Fertilizer production	1333	168	3041	790	3606	56	111	113	6573	31	297	474	3425	64	528	28
Direct nitrous oxide	1427	1274	939	668	709	325	325	556	3577	119	90	284	2151	331	400	162
Indirect nitrous oxide	992	965	569	506	455	246	246	421	2173	90	55	215	1540	251	267	123
Pesticides	21	54	39	70	103	8	8	54	2629	468	28	41	54	33	13	1
Irrigation infraestructure	311	311	311	311	311	311	311	311	311	311	311	311	311	311	311	311
Irrigation energy	702	702	649	649	649	649	649	649	930	930	158	158	1088	1088	246	246
Methane	0	0	160	118	135	63	168	93	135	106	19	0	0	177	110	130
Greenhouse	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157	5157
Nursery	200	200	115	115	182	182	121	121	50	65	484	484	151	151	12	11
Carbon	-1369	-2855	0	-722	-233	-1148	-731	-1735	-82	-607	0	-636	-2334	-1152	-410	-738
Total	12077	9272	11749	8446	12113	6642	7279	6464	23089	8506	7309	7136	13653	8150	7461	6118
Product-based emissions (g CO_2e/kg)																
Machinery production	4	4	0	0	0	0	0	0	1	2	1	1	1	1	1	1
Fuel production	7	9	1	1	1	1	1	1	2	3	2	3	2	2	3	3
Fuel use	46	56	5	3	5	7	3	4	13	19	11	18	13	9	16	20
Fertilizer production	23	4	32	11	45	2	1	2	82	0	10	16	49	1	31	1
Direct nitrous oxide	25	27	10	10	9	9	3	10	45	2	3	9	31	6	24	8
Indirect nitrous oxide	17	20	6	7	6	7	2	8	27	1	2	7	22	4	16	6
Pesticides	0	1	0	1	1	0	0	1	33	7	1	1	1	1	1	0
Irrigation infraestructure	5	6	3	4	4	8	3	6	4	5	10	10	4	5	18	16
Irrigation energy	12	15	7	9	8	17	6	12	12	15	5	5	16	18	14	12
Methane	0	0	2	2	2	2	2	2	2	2	1	0	0	3	6	6
Greenhouse	90	107	54	74	64	138	52	94	64	82	172	172	74	86	303	258
Nursery	3	4	1	2	2	5	1	2	1	1	16	16	2	3	1	1
Carbon	-24	-59	0	-10	-3	-31	-7	-32	-1	-10	0	-21	-33	-19	-24	-37
Total	210	193	124	121	151	177	73	118	288	135	244	238	195	136	439	306
Coproduct	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix A4. Supplementary data of Study 4

Table A4.1 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) citrus crops in Spain. Data refer to 1 ha and year unless otherwise stated

	Tangerine 1		Tangerine 2		Orange 1		Orange 2		Orange 3	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs										
Drip (%)	100%	100%	0%	0%	100%	100%	100%	100%	0%	0%
Surface (%)	0%	0%	100%	100%	0%	0%	0%	0%	100%	100%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Water (m3)	6000	6000	10000	6000	5200	5200	5200	10000	10000	10000
Electricity (KWh)	1896	1896	3160	1896	1643	1643	1643	3160	3160	3160
Presence of cover crops (%)	0%	100%	0%	100%	0%	100%	0%	0%	0%	20%
Machinery use (hours)	37	12	14	16	47	34	12	19	9	8
Fuel consumption (liters)	285	155	107	193	377	177	138	214	63	77
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	311	0	230	0	93	0	244	0	540	0
Mineral phosphorus (kg P2O5)	55	0	71	0	0	0	120	0	390	0
Mineral potassium (kg K2O)	152	0	71	0	88	0	60	0	315	0
Manure (Mg)	0.00	10.00	0.00	10.00	0.00	0.00	0.00	30.00	0.00	27.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	10.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	36.70	0.00	36.70	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	120.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	2777	4042	2403	4562	0	2718	1972	4241	0	4364
Total nitrogen inputs (kg)	371	167	282	179	93	93	287	238	540	232
Synthetic pesticides (kg active matter)	18.9	0.0	6.1	0.0	26.8	0.0	44.1	0.0	23.3	0.0
Sulphur (kg)	0	0	0	500	0	0	0	0	0	0
Copper (kg)	0	0	0	0	0	0	0	0	0	0
Paraffin (kg)	1	0	1	10	1	0	12	0	0	24
Natural pesticides (kg)	0	0	0	0	0	15	0	0	0	10
Production										

Yield (Mg)	42.75	16.00	37.00	24.00	30.00	15.00	50.00	33.00	50.00	33.30
Cover crop (Mg dry matter)	0.00	3.01	0.00	3.01	0.00	3.01	0.00	0.00	0.00	0.60
Residue yield (Mg)	7.54	2.82	6.53	4.23	3.21	1.61	5.36	3.53	5.36	3.57
Residue destiny										
Open burning (Mg)	0.00	0.00	0.00	0.00	2.51	0.00	0.00	0.00	4.18	0.00
Soil incorporation (Mg)	5.88	2.20	5.09	3.30	0.00	1.25	4.18	2.76	0.00	2.78
Coproduct (Mg)	1.66	0.62	1.44	0.93	0.71	0.35	1.18	0.78	1.18	0.78
Allocation to coproduct (%)	3%	3%	3%	3%	2%	2%	2%	2%	2%	2%
Soil emissions										
Direct nitrous oxide (kg N2O)	3.85	1.74	4.48	2.84	0.96	0.97	2.97	2.47	8.57	3.69
Indirect nitrous oxide (kg N2O)	2.43	1.32	1.86	1.40	0.58	0.73	1.87	1.87	3.39	1.83
Methane (kg CH4)	0.00	0.00	0.00	0.00	6.77	0.00	0.00	0.00	11.28	0.00
Carbon (kg C)	-580	-885	-467	-1043	0	-481	-601	-1026	265	-1260

Table A4.2 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) fruits in Spain. Data refer to 1 ha and year unless otherwise stated

	Apple 1		Apple 2		Apple 3		Apple 4		Pear 1		Pear 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs												
Drip (%)	0%	0%	0%	0%	0%	0%	100%	100%	0%	0%	0%	0%
Surface (%)	0%	0%	0%	0%	100%	100%	0%	0%	100%	100%	100%	100%
Rainfed (%)	100%	100%	100%	100%	0%	0%	0%	0%	0%	0%	0%	0%
Water (m3)	0	0	0	0	10000	10000	3800	3800	10000	10000	10000	10000
Electricity (KWh)	0	0	0	0	3160	3160	1201	1201	3160	3160	3160	3160
Presence of cover crops (%)	0%	100%	0%	100%	0%	0%	0%	100%	0%	100%	0%	100%
Machinery use (hours)	46	53	17	54	17	33	31	23	17	15	20	33
Fuel consumption (liters)	178	279	192	505	155	339	282	196	174	166	188	353
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	38	0	125	0	125	0	103	0	164	0	117	0
Mineral phosphorus (kg P2O5)	25	0	50	0	50	0	25	0	66	0	63	0
Mineral potassium (kg K2O)	38	0	150	0	150	2	118	150	320	0	89	0
Manure (Mg)	0.00	10.00	0.00	15.00	0.00	26.00	15.00	7.00	0.00	2.25	0.00	15.00

Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	65.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	65.00
Other organic fertilizers (kg)	0.00	1175.00	0.00	0.00	0.00	0.00	30.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	0	3638	4055	5635	4055	6423	1473	4784	2759	2637	2980	4980
Total nitrogen inputs (kg)	38	141	188	277	188	287	209	116	209	61	166	268
Synthetic pesticides (kg active matter)	1.2	0.0	24.0	0.0	24.0	0.0	19.7	0.0	12.4	0.0	23.0	0.0
Sulphur (kg)	0	0	0	75	0	178	80	25	15	0	0	2
Copper (kg)	0	0	7	35	7	10	7	7	9	0	9	7
Paraffin (kg)	0	0	0	0	0	0	1	1	44	0	2	0
Natural pesticides (kg)	0	0	3	64	3	1	0	0	1	1	0	20
Production												
Yield (Mg)	22.30	7.00	43.00	25.00	43.00	45.00	40.00	22.00	32.40	5.00	35.00	20.00
Cover crop (Mg dry matter)	0.00	3.01	0.00	3.01	0.00	0.00	0.00	3.01	0.00	3.01	0.00	3.01
Residue yield (Mg)	5.69	1.79	10.98	6.38	10.98	11.49	10.21	5.62	7.60	1.17	8.21	4.69
Residue destiny												
Open burning (Mg)	4.44	0.00	0.00	0.00	0.00	0.00	7.97	0.00	0.00	0.00	0.00	0.00
Soil incorporation (Mg)	0.00	1.39	8.56	4.98	8.56	8.96	0.00	4.38	5.93	0.91	6.40	3.66
Coproduct (Mg)	1.25	0.39	2.42	1.40	2.42	2.53	2.25	1.24	1.67	0.26	1.81	1.03
Allocation to coproduct (%)	6%	6%	6%	6%	6%	6%	6%	6%	5%	5%	5%	5%
Soil emissions												
Direct nitrous oxide (kg N_2O)	0.05	0.18	0.24	0.35	2.99	4.56	2.16	1.21	3.32	0.97	2.63	4.25
Indirect nitrous oxide (kg N_2O)	0.24	1.10	1.28	2.18	1.28	2.26	1.48	0.91	1.38	0.48	1.12	2.10
Methane (kg CH_4)	11.99	0.00	0.00	0.00	0.00	0.00	21.51	0.00	0.00	0.00	0.00	0.00
Carbon (kg C)	0	-762	-1235	-1370	-1235	-1956	-449	-1111	-575	-457	-642	-1170

Table A4.2 (Continued) Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) fruits in Spain. Data refer to 1 ha and year unless otherwise stated

	Plum		Grape		Peach		Apricot		Fig 1		Fig 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs												
Drip (%)	0%	0%	100%	100%	100%	100%	100%	100%	0%	0%	100%	100%

Surface (%)	100%	100%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	100%	100%	0%	0%
Water (m3)	10000	10000	3100	3100	6300	6300	1500	1500	0	0	2400	2400
Electricity (KWh)	3160	3160	979	979	1991	1991	474	474	0	0	758	758
Presence of cover crops (%)	0%	100%	0%	0%	0%	100%	0%	100%	0%	50%	50%	100%
Machinery use (hours)	19	33	11	9	29	27	19	35	5	5	26	10
Fuel consumption (liters)	202	213	78	119	283	301	232	162	53	53	256	116
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	90	0	113	0	80	0	138	0	0	0	0	0
Mineral phosphorus (kg P2O5)	90	0	113	0	160	0	114	0	0	0	0	0
Mineral potassium (kg K2O)	90	0	113	0	200	0	150	0	0	0	0	200
Manure (Mg)	0.00	0.00	0.00	8.00	0.00	0.00	0.00	15.00	0.00	0.00	10.00	13.80
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	46.00	0.00	0.00	0.00	65.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	5000.00	0.00	334.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	2556	5197	0	817	3260	3979	1050	4280	0	1143	4976	4780
Total nitrogen inputs (kg)	133	270	113	66	121	100	150	149	0	21	156	164
Synthetic pesticides (kg active matter)	3.9	0.0	12.3	0.0	18.2	0.0	4.0	0.0	0.0	0.0	11.8	0.0
Sulphur (kg)	0	2	35	39	0	0	0	3	0	0	4	0
Copper (kg)	7	0	4	0	0	2	6	10	0	0	0	0
Paraffin (kg)	10	0	0	0	10	0	25	20	0	0	0	0
Natural pesticides (kg)	0	0	0	0	0	1	0	0	0	0	1	0
Production												
Yield (Mg)	20.00	21.00	16.90	11.25	50.00	30.00	9.60	7.20	1.10	0.75	17.00	8.00
Cover crop (Mg dry matter)	0.00	3.01	0.00	0.00	0.00	3.01	0.00	3.01	0.00	1.51	1.67	3.01
Residue yield (Mg)	6.81	7.15	5.82	3.88	8.63	5.18	2.79	2.10	0.54	0.37	8.40	3.95
Residue destiny												
Open burning (Mg)	0.00	0.00	4.54	3.02	0.00	0.00	0.00	0.00	0.42	0.00	0.00	0.00
Soil incorporation (Mg)	5.31	5.57	0.00	0.00	6.73	4.04	2.18	1.63	0.00	0.29	6.55	3.08
Coproduct (Mg)	1.50	1.57	1.28	0.85	1.90	1.14	0.61	0.46	0.12	0.08	1.85	0.87
Allocation to coproduct (%)	6%	6%	4%	4%	2%	2%	3%	3%	3%	3%	3%	3%
Soil emissions												
Direct nitrous oxide (kg N2O)	2.11	4.29	1.17	0.68	1.26	1.03	1.55	1.54	0.00	0.03	1.62	1.70

Indirect nitrous oxide (kg N2O)	0.90	2.12	0.71	0.52	0.83	0.78	0.96	1.17	0.00	0.16	1.23	1.29
Methane (kg CH4)	0.00	0.00	12.26	8.16	0.00	0.00	0.00	0.00	1.14	0.00	0.00	0.00
Carbon (kg C)	-778	-1236	265	-249	-728	-865	-320	-957	0	-175	-1342	-1109

Table A4.3 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) subtropical fruit crops in Spain. Data refer to 1 ha and year unless otherwise stated

	Avocado 1		Avocado 2		Mango		Banana 1		Banana 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs										
Drip (%)	100%	100%	0%	0%	100%	100%	100%	100%	100%	100%
Surface (%)	0%	0%	100%	100%	0%	0%	0%	0%	0%	0%
Rainfed (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Water (m3)	5000	5000	5000	5000	4500	4000	9500	9500	8500	8500
Electricity (KWh)	1580	1580	1580	1580	1422	1264	3002	3002	2686	2686
Presence of cover crops (%)	100%	100%	100%	100%	0%	100%	0%	66%	100%	100%
Machinery use (hours)	8	35	33	22	32	55	9	0	19	42
Fuel consumption (liters)	26	103	12	8	227	258	70	0	141	100
Mulching plastic (kg)	0	0	0	0	0	0	0	110	0	0
Mineral nitrogen (kg N)	116	0	84	0	0	0	318	0	564	0
Mineral phosphorus (kg P2O5)	0	0	96	0	0	0	92	0	130	0
Mineral potassium (kg K2O)	110	0	42	0	50	0	750	0	1201	0
Manure (Mg)	0.00	9.00	0.00	0.00	0.00	0.00	0.00	10.00	4.00	7.50
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	2500.00	0.00	0.00	0.00	213.67	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	7211	3593	2023	2023	0	2084	0	2175	2415	2758
Total nitrogen inputs (kg)	244	236	119	35	0	55	318	83	627	88
Synthetic pesticides (kg active matter)	0.3	0.0	0.0	0.0	7.4	0.0	4.2	0.0	24.6	0.0
Sulphur (kg)	0	0	0	0	6	16	0	0	0	0
Copper (kg)	0	0	0	0	0	0	0	0	0	0
Paraffin (kg)	0	0	0	0	0	0	0	0	0	0

Natural pesticides (kg)	0	10	0	0	0	0	0	0	0	10
Production										
Yield (Mg)	50.00	10.00	50.00	21.00	22.50	18.00	9.00	36.00	24.00	45.00
Cover crop (Mg dry matter)	3.35	3.01	3.35	3.01	0.00	3.01	0.00	1.99	3.35	3.01
Residue yield (Mg)	14.09	2.82	14.09	5.92	5.72	4.58	2.03	8.10	5.40	10.13
Residue destiny										
Open burning (Mg)	0.00	2.20	10.99	4.62	4.46	3.57	2.03	8.10	5.40	10.13
Soil incorporation (Mg)	10.99	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coproduct (Mg)	3.10	0.62	3.10	1.30	1.26	1.01	0.00	0.00	0.00	0.00
Allocation to coproduct (%)	1%	1%	1%	1%	2%	2%	2%	2%	2%	2%
Soil emissions										
Direct nitrous oxide (kg N2O)	2.53	2.44	1.89	0.56	0.00	0.57	3.30	0.86	6.50	0.91
Indirect nitrous oxide (kg N2O)	1.73	1.85	0.80	0.28	0.00	0.43	2.00	0.65	4.04	0.69
Methane (kg CH4)	0.00	5.93	29.67	12.46	12.05	9.64	5.47	21.87	14.58	27.34
Carbon (kg C)	-1850	-748	-270	-270	265	-288	265	-434	-389	-494

Table A4.4 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) treenuts in Spain. Data refer to 1 ha and year unless otherwise stated

	Almond 1		Almond 2		Almond 3		Hazelnut 1		Hazelnut 2		Carob	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs												
Drip (%)	0%	0%	0%	0%	100%	100%	100%	100%	100%	100%	0%	0%
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	100%	100%	100%	100%	0%	0%	0%	0%	0%	0%	100%	100%
Water (m3)	0	0	0	0	800	500	210	210	210	210	0	0
Electricity (KWh)	0	0	0	0	253	158	66	66	66	66	0	0
Presence of cover crops (%)	0%	0%	0%	0%	0%	0%	0%	100%	0%	0%	100%	100%
Machinery use (hours)	20	29	9	8	10	17	65	95	131	107	1	6
Fuel consumption (liters)	241	401	142	122	113	122	172	317	403	490	0	79
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	12	0	8	0	186	0	30	0	16	0	0	0
Mineral phosphorus (kg P2O5)	6	0	8	0	72	0	30	0	48	0	0	0

Mineral potassium (kg K2O)	15	0	8	0	72	0	43	0	96	21	0	0
Manure (Mg)	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	2.63	0.00	0.00	0.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	250.00	0.00	650.00	0.00	0.00	0.00	325.00	0.00	0.00
Total carbon inputs (kg)	0	98	0	25	0	64	0	2023	257	32	2023	2023
Total nitrogen inputs (kg)	12	7	8	8	186	20	30	20	34	10	20	20
Synthetic pesticides (kg active matter)	0.3	0.0	0.0	0.0	6.1	0.0	1.1	0.0	2.5	0.0	0.0	0.0
Sulphur (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Copper (kg)	0	0	0	0	0	0	0	0	0	0	0	0
Paraffin (kg)	0	0	0	0	9	9	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	0	0	4	0	0	0	10	0	0
Production												
Yield (Mg)	0.80	1.04	0.45	0.42	5.00	3.75	1.20	1.53	1.80	1.30	2.53	2.64
Cover crop (Mg dry matter)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.01	0.00	0.00	3.01	3.01
Residue yield (Mg)	1.25	1.63	0.71	0.66	7.87	5.90	1.53	1.95	2.30	1.66	3.50	3.65
Residue destiny												
Open burning (Mg)	0.98	1.27	0.55	0.52	6.14	4.60	1.19	1.52	1.79	1.29	2.73	2.85
Soil incorporation (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coproduct (Mg)	0.28	0.36	0.16	0.15	1.73	1.30	0.34	0.43	0.50	0.36	0.77	0.80
Allocation to coproduct (%)	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	33%	33%
Soil emissions												
Direct nitrous oxide (kg N2O)	0.02	0.01	0.01	0.01	1.93	0.20	0.31	0.21	0.36	0.10	0.03	0.03
Indirect nitrous oxide (kg N2O)	0.08	0.06	0.05	0.06	1.17	0.15	0.19	0.16	0.24	0.08	0.16	0.16
Methane (kg CH4)	2.63	3.44	1.49	1.39	16.57	12.42	3.22	4.11	4.83	3.49	7.36	7.68
Carbon (kg C)	0	-30	0	-7	0	-19	0	-270	-78	-10	-270	-270

Table A4.5 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) vineyards in Spain. Data refer to 1 ha and year unless otherwise stated

	Vineyard 1		Vineyard 2		Vineyard 3		Vineyard 4		Vineyard 5		Vineyard 6		Vineyard 7	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org

Inputs														
Drip (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Water (m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Electricity (KWh)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Presence of cover crops (%)	0%	100%	0%	0%	0%	0%	0%	0%	0%	0%	100%	0%	0%	100%
Machinery use (hours)	10	7	44	60	9	10	17	15	18	18	3	12	22	6
Fuel consumption (liters)	114	104	476	755	145	160	162	150	126	126	52	183	263	77
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	8	0	26	0	23	0	0	0	0	0	0	0	0	0
Mineral phosphorus (kg P2O5)	12	0	120	0	45	0	0	0	0	0	0	0	0	0
Mineral potassium (kg K2O)	24	0	138	0	68	0	0	0	0	0	0	0	0	0
Manure (Mg)	0.00	2.10	0.00	2.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.00	3.14	5.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	150.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total carbon inputs (kg)	0	2199	0	196	0	457	512	549	0	0	2023	294	263	2513
Total nitrogen inputs (kg)	8	43	26	14	23	6	7	7	0	0	20	21	35	55
Synthetic pesticides (kg active matter)	1.7	0.0	19.4	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sulphur (kg)	27	50	109	18	25	25	0	2	6	6	50	25	8	27
Copper (kg)	0	1	9	5	0	0	0	1	1	1	0	0	0	0
Paraffin (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Production														
Yield (Mg)	4.00	2.70	7.30	5.00	4.70	5.00	5.60	6.00	5.50	5.50	12.00	10.80	5.50	5.50
Cover crop (Mg dry matter)	0.00	3.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.01	0.00	0.00	3.01
Residue yield (Mg)	0.99	0.67	1.80	1.24	1.16	1.24	1.38	1.48	1.36	1.36	2.96	2.67	1.36	1.36
Residue destiny														
Open burning (Mg)	0.77	0.52	1.41	0.96	0.91	0.00	0.00	0.00	1.06	1.06	2.31	2.08	1.06	1.06
Soil incorporation (Mg)	0.00	0.00	0.00	0.00	0.00	0.96	1.08	1.16	0.00	0.00	0.00	0.00	0.00	0.00
Coproduct (Mg)	0.22	0.15	0.40	0.27	0.26	0.27	0.30	0.33	0.30	0.30	0.65	0.59	0.30	0.30
Allocation to coproduct (%)	6%	6%	6%	6%	6%	6%	6%	6%	6%	6%	6%	6%	6%	6%

Soil emissions														
Direct nitrous oxide (kg N2O)	0.01	0.05	0.03	0.02	0.03	0.01	0.01	0.01	0.00	0.00	0.03	0.03	0.04	0.07
Indirect nitrous oxide (kg N2O)	0.05	0.34	0.17	0.11	0.14	0.05	0.05	0.06	0.00	0.00	0.16	0.17	0.27	0.43
Methane (kg CH4)	2.08	1.40	3.80	2.60	2.44	0.00	0.00	0.00	2.86	2.86	6.24	5.62	2.86	2.86
Carbon (kg C)	0	-323	0	-60	0	-139	-156	-167	265	0	-270	-90	-80	-419

Table A4.6 Main characteristics of the life cycle inventory of the studied conventional (Con) and organic (Org) olives in Spain. Data refer to 1 ha and year unless otherwise stated

	Olive 1		Olive 2		Olive 3		Olive 4		Olive 5		Olive 6		Olive 7	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Inputs														
Drip (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Surface (%)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Rainfed (%)	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Water (m3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Electricity (KWh)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Presence of cover crops (%)	0%	100%	0%	100%	0%	0%	50%	100%	0%	100%	0%	100%	0%	100%
Machinery use (hours)	29	21	29	30	29	23	28	31	46	37	28	35	29	22
Fuel consumption (liters)	279	230	83	137	226	176	98	38	130	80	30	95	92	44
Mulching plastic (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mineral nitrogen (kg N)	10	0	51	0	10	0	149	0	107	0	56	0	86	0
Mineral phosphorus (kg P2O5)	15	0	45	0	20	0	14	0	23	0	56	0	0	6
Mineral potassium (kg K2O)	30	0	64	0	20	0	23	0	38	0	56	0	0	8
Manure (Mg)	10.00	15.00	0.00	5.00	0.00	0.00	0.00	0.00	0.00	3.00	0.00	5.00	0.00	0.00
Slurry (Mg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cover crop seeds (kg)	0.00	0.00	0.00	150.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other organic fertilizers (kg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1500.00	0.00	0.00	0.00	150.00	0.00	1250.00
Total carbon inputs (kg)	840	3517	421	2809	0	0	1012	2591	0	2726	0	3206	339	2922
Total nitrogen inputs (kg)	70	114	58	106	10	0	159	72	107	48	56	70	91	201
Synthetic pesticides (kg active matter)	1.4	0.0	2.3	0.0	1.2	0.0	2.6	0.0	1.6	0.0	3.3	0.0	0.8	0.0
Sulphur (kg)	0	0	0	0	0	0	0	0	300	0	0	0	0	0

Copper (kg)	8	0	14	14	0	0	7	7	14	7	14	7	11	7
Paraffin (kg)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Natural pesticides (kg)	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Production														
Yield (Mg)	2.75	2.00	3.60	3.15	1.50	1.50	3.00	3.60	7.80	3.50	5.00	5.80	2.90	3.20
Cover crop (Mg dry matter)	0.00	3.01	0.00	3.01	0.00	0.00	1.51	3.01	0.00	3.01	0.00	3.01	0.00	3.01
Residue yield (Mg)	0.89	0.64	1.16	1.01	0.48	0.48	0.97	1.16	2.51	1.13	1.61	1.87	0.93	1.03
Residue destiny														
Open burning (Mg)	0.69	0.00	0.00	0.00	0.38	0.38	0.75	0.00	1.96	0.00	1.26	0.00	0.00	0.00
Soil incorporation (Mg)	0.00	0.50	0.90	0.79	0.00	0.00	0.00	0.90	0.00	0.88	0.00	1.46	0.73	0.80
Coproduct (Mg)	0.19	0.14	0.26	0.22	0.11	0.11	0.21	0.26	0.55	0.25	0.35	0.41	0.21	0.23
Allocation to coproduct (%)	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%
Soil emissions														
Direct nitrous oxide (kg N2O)	0.09	0.14	0.07	0.13	0.01	0.00	0.20	0.09	0.13	0.06	0.07	0.09	0.11	0.25
Indirect nitrous oxide (kg N2O)	0.53	0.89	0.37	0.83	0.07	0.00	1.01	0.57	0.67	0.38	0.35	0.55	0.58	1.58
Methane (kg CH4)	1.86	0.00	0.00	0.00	1.02	1.02	2.03	0.00	5.29	0.00	3.39	0.00	0.00	0.00
Carbon (kg C)	-256	-725	137	-509	0	0	-135	-443	0	-484	265	-630	-103	-544

Table A4.7 Global warming potential of organic and conventional Spanish citrus crops for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Tangerine 1		Tangerine 2		Orange 1		Orange 2		Orange 3	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)										
Machinery production	67	30	25	37	92	33	30	49	15	19
Fuel production	123	67	46	84	163	77	60	93	27	34
Fuel use	768	417	288	521	1016	476	371	578	169	209
Fertilizer production	1973	45	1212	45	747	51	1420	113	3222	101
Direct nitrous oxide	1146	518	1335	845	287	289	886	736	2554	1099
Indirect nitrous oxide	723	392	553	418	174	219	557	558	1011	544
Pesticides	343	0	63	103	251	115	642	0	366	38
Irrigation infraestructure	311	311	34	311	311	311	311	34	34	34
Irrigation energy	1053	1053	1755	1053	913	913	913	1755	1755	1755

Methane	0	0	0	0	169	0	0	0	282	0
Greenhouse	0	0	0	0	0	0	0	0	0	0
Nursery	633	272	608	287	618	330	503	404	1083	316
Carbon	-1064	-1622	-855	-1912	0	-882	-1101	-1882	487	-2310
Total	6076	1482	5066	1792	4794	1930	4592	2438	11093	1840
Product-based emissions (g CO2e/kg)										
Machinery production	2	2	1	2	3	2	1	1	0	1
Fuel production	3	4	1	3	5	5	1	3	1	1
Fuel use	18	25	8	21	33	31	7	17	3	6
Fertilizer production	45	3	32	2	24	3	28	3	63	3
Direct nitrous oxide	26	32	35	34	9	19	17	22	50	32
Indirect nitrous oxide	16	24	15	17	6	14	11	16	20	16
Pesticides	8	0	2	4	8	7	13	0	7	1
Irrigation infraestructure	7	19	1	13	10	20	6	1	1	1
Irrigation energy	24	64	46	43	30	59	18	52	34	51
Methane	0	0	0	0	6	0	0	0	6	0
Greenhouse	0	0	0	0	0	0	0	0	0	0
Nursery	14	17	16	12	20	21	10	12	21	9
Carbon	-24	-99	-23	-78	0	-57	-22	-56	10	-68
Total	139	90	133	73	156	126	90	72	217	54
Coproduct	4	2	3	2	4	3	2	2	5	1

Table A4.8 Global warming potential of organic and conventional Spanish fruits for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Apple 1		Apple 2		Apple 3		Apple 4		Pear 1		Pear 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)												
Machinery production	35	62	38	130	38	77	73	53	41	38	47	83
Fuel production	78	122	83	219	67	147	122	85	76	72	81	153
Fuel use	476	751	517	1363	417	915	761	529	470	448	507	951
Fertilizer production	221	166	789	69	789	100	710	286	1207	8	902	69

Direct nitrous oxide	14	53	71	104	891	1359	645	360	988	289	784	1267
Indirect nitrous oxide	70	329	382	648	382	673	440	273	412	143	333	627
Pesticides	15	2	243	262	243	72	180	24	193	5	283	53
Irrigation infraestructure	0	0	0	0	34	34	311	311	34	34	34	34
Irrigation energy	0	0	0	0	1755	1755	667	667	1755	1755	1755	1755
Methane	300	0	0	0	0	0	538	0	0	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	95	9	-29	38	250	182	318	94	420	322	395	333
Carbon	0	-1396	-2264	-2511	-2264	-3586	-822	-2036	-1054	-837	-1177	-2145
Total	1560	704	643	1514	2595	1715	4443	831	4544	2277	3943	3181
Product-based emissions (g CO2e/kg)												
Machinery production	1	8	1	5	1	2	2	2	1	7	1	4
Fuel production	3	16	2	8	1	3	3	4	2	14	2	7
Fuel use	20	101	11	51	9	19	18	23	14	86	14	45
Fertilizer production	9	22	17	3	17	2	17	12	36	2	25	3
Direct nitrous oxide	1	7	2	4	20	28	15	15	29	55	21	60
Indirect nitrous oxide	3	44	8	24	8	14	10	12	12	27	9	30
Pesticides	1	0	5	10	5	2	4	1	6	1	8	3
Irrigation infraestructure	0	0	0	0	1	1	7	13	1	7	1	2
Irrigation energy	0	0	0	0	38	37	16	29	52	335	48	84
Methane	13	0	0	0	0	0	13	0	0	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	4	1	-1	1	5	4	8	4	12	61	11	16
Carbon	0	-188	-50	-95	-50	-75	-19	-87	-31	-160	-32	-102
Total	66	95	14	57	57	36	105	36	134	435	107	152
Coproduct	4	6	1	3	3	2	6	2	6	21	5	7

Table A4.8 (Continued) Global warming potential of organic and conventional Spanish fruits for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Plum		Grape		Peach		Apricot		Fig 1		Fig 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)												

Machinery production	46	43	24	25	68	68	50	37	10	10	61	25
Fuel production	87	93	34	52	122	130	100	71	23	23	111	50
Fuel use	544	574	211	321	763	811	626	435	143	143	691	314
Fertilizer production	576	558	720	67	680	13	880	56	0	0	38	398
Direct nitrous oxide	629	1278	348	204	374	308	463	460	0	8	483	507
Indirect nitrous oxide	269	633	211	155	246	233	286	348	0	48	366	384
Pesticides	78	2	162	8	182	11	75	38	0	0	124	0
Irrigation infraestructure	34	34	311	311	311	311	311	311	0	0	311	311
Irrigation energy	1755	1755	544	544	1106	1106	263	263	0	0	421	421
Methane	0	0	307	204	0	0	0	0	29	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	593	530	485	282	620	480	269	49	25	-14	52	56
Carbon	-1427	-2267	487	-456	-1334	-1587	-586	-1755	0	-321	-2461	-2034
Total	3184	3232	3937	1780	3138	1885	2737	315	239	-104	197	432
Product-based emissions (g CO2e/kg)												
Machinery production	2	2	1	2	1	2	5	5	9	12	3	3
Fuel production	4	4	2	4	2	4	10	10	20	30	6	6
Fuel use	26	26	12	27	15	27	63	58	126	184	39	38
Fertilizer production	27	25	41	6	13	0	89	8	0	0	2	48
Direct nitrous oxide	30	57	20	17	7	10	47	62	0	10	28	61
Indirect nitrous oxide	13	28	12	13	5	8	29	47	0	62	21	47
Pesticides	4	0	9	1	4	0	8	5	0	0	7	0
Irrigation infraestructure	2	2	18	26	6	10	31	42	0	0	18	38
Irrigation energy	83	79	31	46	22	36	27	35	0	0	24	51
Methane	0	0	17	17	0	0	0	0	25	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	28	24	27	24	12	16	27	7	22	-18	3	7
Carbon	-67	-102	28	-39	-26	-52	-59	-236	0	-415	-140	-247
Total	150	145	223	151	62	62	275	42	210	-135	11	52
Coproduct	9	9	10	7	1	1	10	1	6	-4	0	2

Table A4.9 Global warming potential of organic and conventional Spanish subtropical fruits for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Avocado 1		Avocado 2		Mango		Banana 1		Banana 2	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)										
Machinery production	5	24	1	1	57	62	15	0	35	33
Fuel production	11	45	7	4	98	112	30	0	61	45
Fuel use	69	275	30	20	611	695	188	0	379	266
Fertilizer production	1014	516	531	0	134	34	2746	38	6776	28
Direct nitrous oxide	754	728	563	166	0	170	983	257	1937	270
Indirect nitrous oxide	517	551	239	82	0	129	596	195	1203	205
Pesticides	5	17	0	0	195	4	39	1	230	16
Irrigation infraestructure	311	311	144	144	311	311	311	311	311	311
Irrigation energy	878	878	878	878	790	702	1667	1667	1492	1492
Methane	0	148	742	312	301	241	137	547	365	683
Greenhouse	0	0	0	0	0	0	0	0	0	0
Nursery	26	294	375	248	490	339	0	0	0	0
Carbon	-3391	-1371	-494	-494	487	-529	487	-795	-713	-905
Total	199	2463	3243	1454	3567	2345	7241	2620	12189	2657
Product-based emissions (g CO2e/kg)										
Machinery production	0	2	0	0	2	3	2	0	1	1
Fuel production	0	4	0	0	4	6	3	0	2	1
Fuel use	1	27	1	1	27	38	20	0	15	6
Fertilizer production	20	51	10	0	6	2	298	1	276	1
Direct nitrous oxide	15	72	11	8	0	9	107	7	79	6
Indirect nitrous oxide	10	54	5	4	0	7	65	5	49	4
Pesticides	0	2	0	0	9	0	4	0	9	0
Irrigation infraestructure	6	31	3	7	14	17	34	8	13	7
Irrigation energy	17	86	17	41	34	38	181	45	61	32
Methane	0	15	15	15	13	13	15	15	15	15
Greenhouse	0	0	0	0	0	0	0	0	0	0
Nursery	1	29	7	12	21	19	0	0	0	0
Carbon	-67	-135	-10	-23	21	-29	53	-22	-29	-20
Total	4	243	64	68	156	128	785	71	496	58
Coproduct	0	4	1	1	3	2	19	2	12	1

Table A4.10 Global warming potential of organic and conventional Spanish treenuts for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Almond 1		Almond 2		Almond 3		Hazelnut 1		Hazelnut 2		Carob	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)												
Machinery production	53	78	26	22	23	24	33	60	78	86	0	14
Fuel production	104	174	62	53	49	53	76	139	178	215	0	34
Fuel use	649	1080	384	329	304	328	459	847	1079	1316	1	213
Fertilizer production	76	4	48	27	1002	71	199	0	182	72	0	0
Direct nitrous oxide	4	3	3	3	575	60	93	62	106	30	7	7
Indirect nitrous oxide	22	16	14	18	348	46	56	47	73	23	47	47
Pesticides	4	0	0	0	71	34	17	0	29	17	0	0
Irrigation infraestructure	0	0	0	0	311	311	311	311	311	311	0	0
Irrigation energy	0	0	0	0	140	88	37	37	37	37	0	0
Methane	66	86	37	35	414	311	81	103	121	87	184	192
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	115	171	67	56	343	203	190	175	275	304	-31	2
Carbon	0	-55	0	-14	0	-36	0	-494	-144	-18	-494	-494
Total	1115	1583	652	540	3708	1590	1577	1319	2363	2508	-229	75
Product-based emissions (g CO2e/kg)												
Machinery production	63	71	54	50	4	6	26	37	41	63	0	4
Fuel production	124	158	130	119	9	13	60	86	94	157	0	9
Fuel use	773	985	807	741	58	83	363	525	569	961	0	54
Fertilizer production	90	3	101	62	190	18	157	0	96	52	0	0
Direct nitrous oxide	5	2	6	6	109	15	73	38	56	22	2	2
Indirect nitrous oxide	27	15	30	40	66	12	44	29	38	17	12	12
Pesticides	4	0	0	0	13	9	14	0	15	13	0	0
Irrigation infraestructure	0	0	0	0	59	79	246	193	164	227	0	0
Irrigation energy	0	0	0	0	27	22	29	23	19	27	0	0
Methane	78	78	78	78	78	78	64	64	64	64	49	49
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0

Nursery	137	156	141	126	65	51	150	108	145	222	-8	1
Carbon	0	-50	0	-31	0	-9	0	-307	-76	-13	-131	-125
Total	1327	1444	1371	1215	701	401	1247	818	1246	1830	-61	19
Coproduct	76	83	79	70	40	23	67	44	67	99	-30	9

Table A4.11 Global warming potential of organic and conventional Spanish vineyards for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Vineyard 1		Vineyard 2		Vineyard 3		Vineyard 4		Vineyard 5		Vineyard 6		Vineyard 7	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2e/ha)														
Machinery production	24	20	98	139	26	29	38	32	32	32	9	31	53	15
Fuel production	49	45	206	327	63	69	70	65	54	54	23	79	114	33
Fuel use	307	281	1283	2035	390	431	437	405	339	339	141	494	710	207
Fertilizer production	63	37	545	8	184	0	0	0	0	0	0	11	12	19
Direct nitrous oxide	3	16	10	5	8	2	3	3	0	0	7	8	13	21
Indirect nitrous oxide	15	101	49	33	42	14	16	17	0	0	47	49	81	129
Pesticides	33	12	171	13	5	5	12	4	4	5	10	5	2	5
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	52	35	95	65	61	0	0	0	72	72	156	140	72	72
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	69	-16	291	335	90	38	37	28	134	69	-14	86	116	-46
Carbon	0	-592	0	-109	0	-255	-286	-306	487	0	-494	-164	-147	-768
Total	631	-50	2776	2871	888	334	326	246	1143	592	-67	783	1048	-291
Product-based emissions (g CO2e/kg)														
Machinery production	6	7	13	26	5	6	6	5	5	5	1	3	9	3
Fuel production	12	16	27	61	13	13	12	10	9	9	2	7	19	6
Fuel use	72	98	165	383	78	81	73	63	58	58	11	43	121	35
Fertilizer production	15	13	70	1	37	0	0	0	0	0	0	1	2	3
Direct nitrous oxide	1	6	1	1	2	0	0	0	0	0	1	1	2	4

Indirect nitrous oxide	4	35	6	6	8	3	3	3	0	0	4	4	14	22
Pesticides	8	4	22	3	1	1	2	1	1	1	1	0	0	1
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	12	12	12	12	12	0	0	0	12	12	12	12	12	12
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	16	-5	37	63	18	7	6	4	23	12	-1	7	20	-8
Carbon	0	-206	0	-21	0	-48	-48	-48	83	0	-39	-14	-25	-131
Total	148	-17	358	540	178	63	55	39	195	101	-5	68	179	-50
Coproduct	9	-1	23	34	11	4	3	2	12	6	0	4	11	-3

Table A4.12 Global warming potential of organic and conventional Spanish olives for a 100-year time horizon expressed as kilograms of CO_2e per hectare and year and as grams of CO_2e per kilogram of product

	Olive 1		Olive 2		Olive 3		Olive 4		Olive 5		Olive 6		Olive 7	
	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org	Con	Org
Area-based emissions (kg CO2-eq/ha														
Machinery production	59	50	16	26	46	34	21	7	32	17	7	15	18	8
Fuel production	121	100	37	60	98	77	44	18	58	36	14	43	41	20
Fuel use	752	619	222	366	609	474	263	98	346	211	77	254	245	116
Fertilizer production	116	56	382	48	80	0	698	165	478	11	355	138	163	251
Direct nitrous oxide	26	43	22	40	4	0	59	27	40	18	21	26	34	75
Indirect nitrous oxide	159	267	111	248	19	0	302	168	199	112	104	163	174	471
Pesticides	40	0	64	33	10	0	51	14	101	14	71	14	32	14
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	47	0	0	0	25	25	51	0	132	0	85	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	54	-29	65	-21	65	47	55	-38	81	-41	76	-51	26	-34
Carbon	-469	-1329	252	-933	0	0	-247	-811	0	-887	487	-1155	-189	-996
Total	920	-223	1171	-133	964	666	1313	-352	1508	-509	1322	-552	543	-74

Product-based emissions (g CO2e/kg)														
Machinery production	20	24	4	8	29	22	7	2	4	5	1	2	6	2
Fuel production	42	47	10	18	62	49	14	5	7	10	3	7	13	6
Fuel use	260	295	59	111	386	301	83	26	42	57	15	42	80	35
Fertilizer production	40	27	101	15	51	0	221	43	58	3	68	23	53	75
Direct nitrous oxide	9	20	6	12	2	0	19	7	5	5	4	4	11	22
Indirect nitrous oxide	55	127	29	75	12	0	96	45	24	30	20	27	57	140
Pesticides	14	0	17	10	6	0	16	4	12	4	14	2	11	4
Irrigation infraestructure	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Irrigation energy	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Methane	16	0	0	0	16	16	16	0	16	0	16	0	0	0
Greenhouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nursery	19	-14	17	-6	41	30	18	-10	10	-11	15	-8	8	-10
Carbon	-162	-632	66	-282	0	0	-78	-214	0	-241	93	-189	-62	-296
Total	318	-106	309	-40	611	422	416	-93	184	-138	252	-90	178	-22
Coproduct	16	-5	16	-2	31	22	21	-5	9	-7	13	-5	9	-1