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Dear Editor,

We have addressed suggested minor changes. The abstract was shortened in 200 words (400 length now) and the highlights were modified according to your comments.

We hope the MS is now suitable for publication in this SI of Agriculture, Ecosystems & Environment.

Yours sincerely,

Alberto Sanz-Cobena

***Highlights (for review)**

- Optimized N fertilization and irrigation show a large potential for N_2O mitigation.
- Reduced tillage and crop residues management show a large potential for reducing net GHG emissions.
- CH_4 fluxes from paddies are controlled by management of water table and organic inputs.
- Factors beyond the plot scale may outweigh mitigation measures.
- Training to farmers on the application of practices will overcome barriers for implementation.

Strategies for greenhouse gas emissions mitigation in Mediterranean agriculture: a review

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Abstract

An integrated assessment of the potential of different management practices for mitigating specific components of the total GHG budget (N_2O and CH_4 emissions and C sequestration) of Mediterranean agrosystems was performed in this study. Their suitability regarding both yield and environmental (e.g. nitrate leaching and ammonia volatilization) sustainability, and regional barriers and opportunities for their implementation were also considered. Based on its results best strategies to abate GHG emissions in Mediterranean agro-systems were proposed. Adjusting N fertilization to crop needs in both irrigated and rain-fed systems could reduce N_2O emissions up to 50% compared with a non-adjusted practice. Substitution of N synthetic fertilizers by solid manure can be also implemented in those systems, and may abate N_2O emissions by about 20% under Mediterranean conditions, with additional indirect benefits associated to energy savings and positive effects in crop yields. The use of urease and nitrification inhibitors enhances N use efficiency of the cropping systems and may mitigate N_2O emissions up to 80% and 50%, respectively. The type of irrigation may also have a great mitigation potential in the Mediterranean region. Drip-irrigated systems have on average 80% lower N_2O emissions than sprinkler systems and drip-irrigation combined with optimized fertilization showed a reduction in direct N_2O emissions up to 50%. Methane fluxes have a relatively small contribution to the total GHG budget of Mediterranean crops, which can mostly be controlled by careful management of the water table and organic inputs in paddies. Reduced soil tillage, improved management of crop residues and agro-industry by-products, and cover cropping in orchards, are the most suitable interventions to enhance organic C stocks in Mediterranean agricultural soils. The adoption of the proposed agricultural practices will require farmers training. The global analysis of life cycle emissions associated to irrigation type (drip, sprinkle and furrow) and N fertilization rate (100 and 300 kg N ha^{-1} yr^{-1}) revealed that these factors may outweigh the reduction in GHG emissions beyond the plot scale. The analysis of the impact of some structural changes on top-down mitigation of GHG emissions revealed that 3-15% of N_2O emissions could be suppressed by avoiding food waste at the end-consumer level. A 40% reduction in meat and dairy consumption could reduce GHG emissions by 20 to 30%. Reintroducing the Mediterranean diet (i.e. ~35% intake of animal protein) would therefore result in a significant decrease of GHG emissions from agricultural production systems under Mediterranean conditions.

Keys words: Cropping systems, GHG, Mitigation, Mediterranean climate, review

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1. Introduction

Mediterranean climate, found from 20º latitude onwards, is characterized by having mild winters and warm summers. Precipitation during summer period, when highest temperatures occur, is scarce, so most summer crops require irrigation to achieve worthwhile yields. Mediterranean climate is neither desert climate, nor humid, and three subtypes can be distinguished: humid or rainy Mediterranean (Ln – Seasonal rainfall surplus – higher than 20% of annual PET – potential evapotranspiration); dry Mediterranean, and semiarid Mediterranean (drier than dry Mediterranean climate) (Ln < 20% PET) (Papadakis et al., 1966). Over one half of the area with Mediterranean-type climate worldwide is found in the Mediterranean Sea Basin (Aschmann, 1973), but it is also present in four other regions of the world namely California (USA), Central Chile, the Cape region of South Africa, and South-West Australia (Figure 1).

In the case of Mediterranean agricultural systems, the temporal gap between maximum irradiance and temperature (early summer) and maximum water availability (winter), added to the low organic matter (OM) content of most cropped soils, are important drivers of the typically low productivity of rain-fed crops. On the contrary, irrigated agriculture benefits from the solar radiation and extended frost-free periods to make these areas capable of high crops yields. The different soil conditions between irrigated and rain-fed crops greatly affect soil microbial processes, which control the fluxes of C (carbon dioxide, CO_2; methane, CH_4; organic carbon) and N (nitrous oxide, N_2O; molecular nitrogen, N_2; nitrate, NO_3^-; ammonia, NH_3) in soil.

Pedoclimatic conditions shape soil processes in Mediterranean cropping systems, leading to different N_2O emission patterns compared to temperate soils (Aguilera et al., 2013b). Nitrification and nitrifier-denitrification, and not denitrification, are very often the main pathways leading to emissions of N oxides in rain-fed

Mediterranean cropping system (Sánchez-Martín et al., 2008; Kool et al., 2011; Aguilera et al., 2013b; Vallejo et al., 2014). These two processes are favoured by conditions of soil water content (i.e., water filled pore space, WFPS) under saturation (i.e. 40-60% WFPS). Denitrification may play a predominant role in anaerobic soil microsites (Davidson et al., 1991) in intensively managed and irrigated systems (e.g., Sanz-Cobena et al., 2012; 2014c). Consequently, different cumulative N_2O emissions have been proposed for rain-fed crops (0.7 kg N_2O-N ha^{-1} yr^{-1}) and for e.g., sprinkler irrigated crops in Mediterranean areas (4.4 kg N_2O-N ha^{-1} yr^{-1}) (Cayuela et al., this issue). Thus, the importance and potential for N_2O mitigation and the best mitigation strategy differ greatly depending on the cropping system.

Paddy soils account for 6% of the total CH_4 emissions from Mediterranean agriculture (Tate, 2015). Large CH_4 emissions in these flooded soils are generated through methanogenesis under strict anaerobic conditions and low oxido-reduction potentials (Le Mer and Roger, 2001). On the contrary, aerobic agricultural soils, both rain-fed and irrigated, promote CH_4 oxidation, which is very dependent on management practices such as N fertilization. Agricultural management strategies based on reducing methanogenesis in paddy soils, or enhancement of CH_4 oxidation in aerated soils, are often ignored in Mediterranean agriculture, yet they may contribute substantially to reduce total GHG emissions from these systems.

Increasing the generally low C content of Mediterranean soils is an important GHG mitigation strategy (Robertson et al., 2000), and is also a priority for preventing erosion and improving soil quality.

In this review we have synthesized and analyzed the performance of agronomic GHG mitigation practices in Mediterranean cropping systems aiming to i) decrease soil N_2O emissions; ii) enhance CH_4 oxidation and decrease CH_4 emission rates; iii) enhance soil organic C stocks and iv) reduce or leave unchanged other sources of environmental pollution (e.g. NH_3 volatilization and NO_3^- leaching). The effect on the total GHG budget of the selected strategies was also analyzed to establish an order of priority. The review also includes an assessment of the socioeconomic performance of agronomic measures and constraints to implementation. Finally, we explored the potential of structural measures at the agro-food system scale for reducing GHGs emissions: i) food waste reduction, ii) change in the composition of human diet, particularly in the proportion of animal products, and iii) reconnection between crops and livestock at farm or regional scale for optimization of resource use.

2. Agronomic mitigation measures

2.1. Agronomic practices affecting N_2O emissions

As previously explained, Mediterranean climatic conditions lead to the existence of two main contrasting production systems, rain-fed and irrigated, largely differing in terms of crop management and, consequently, N_2O emission processes. Rain-fed systems, mostly based on winter crops, are characterized by periods with low soil moisture and cold temperatures, thus with decreased soil microbiological activity and N_2O fluxes. The IPCC (2006) has proposed a 1% emission factor (EF, i.e. the percentage of fertilizer N applied that is transformed and emitted back to the atmosphere as N_2O) at Tier 1 (Tier 1 default EF[1] proposed by IPCC, 2006) for N_2O emissions. However, two recent reviews have shown that N_2O emission factors from rain-fed Mediterranean cropping systems are much lower than the default 1% (i.e. Aguilera et al., 2013b; Cayuela et al., this issue). Irrigated systems receive large amounts of water and N inputs which create favorable soil conditions for N_2O

[1] EF_1 for N additions from mineral fertilisers, organic amendments and crop residues, and N mineralised from mineral soil as a result of loss of soil carbon. TABLE 11.1, IPCC 2006 National GHG Inventory Guidelines. Volume IV (AFOLU), Chapter 11.

production. Emission factors in these systems fluctuate greatly according to water management and the type and amount of fertilizer used (e.g., synthetic, solid or liquid manures). Sprinkler irrigated crops led to a N_2O EF similar to those of temperate areas of about 1%; conversely, drip irrigated systems emit at a much lower rate (0.18) (Cayuela et al., this issue).

2.1.1. Nitrogen fertilization

Optimized N fertilizers application (in terms of input rate and time of application), as well as the careful selection of the type of fertilizer used are crucial to reduce N_2O emissions. Synthetic and organic fertilizers are the most widespread sources of environmental N contamination in Mediterranean areas with dense concentration of livestock, due to the losses of N coming from unadjusted fertilizer application (e.g., Sanz-Cobena et al., 2014c). An additional mitigation effect could be achieved by applying already existing N (organic fertilizer) when possible or with the use of nitrification and urease inhibitors.

A. Adjusting N fertilization to crop needs

Recommendations on N application rates, based on a careful estimation of crop needs, aim to achieve optimum yields while reducing N pollution. Reduction of N rates according to soil N availability and crop yield potential may decrease N surpluses and subsequent direct and indirect N_2O emissions, while saving energy and abating other GHG emissions (e.g. associated to manufacturing synthetic fertilizers). Current national emission inventory methods mostly use the 1% Tier 1 EF (IPCC, 2006). However, many studies concluded that the response of direct N_2O emission to N input is non-linear (Philibert et al., 2012; Kim et al., 2013; Shcherbak et al., 2014), and other management factors, as constrained by climate, must be considered in determining N_2O

emissions (Bouwman et al., 2002; Leip et al., 2011; Lesschen et al., 2011; Aguilera et al., 2013a). For example, significant effects of N application timing on N_2O emissions have been reported from cereal crops in Mediterranean countries such as Spain (Abalos et al., 2016). The estimated N_2O mitigation potential, through adjusted fertilization (rate and timing) in Mediterranean agro-ecosystems ranges between 30 and 50% compared to a non-adjusted practice (Table 1).

B. Substituting synthetic fertilizers by organic fertilizers

Differences among fertilizer N sources in N_2O emissions depend on site- and weather-specific conditions (Snyder et al., 2009). Replacing mineral N with organic fertilization provides not only NPK and micronutrients to the soil and crop, but also organic C when using solid fertilizers (i.e., solid manure, composts, etc.), which is highly beneficial in Mediterranean soils with low organic C contents (Aguilera et al., 2013b). In areas were croplands co-exist with livestock farms, using a farm sub-product allows the reuse/recovery of farm products, thus decreasing the volume of waste that needs to be managed, and then avoiding the emission of GHG both in the management of such wastes and in the manufacturing of new synthetic fertilizers. In Mediterranean areas, the efficient use of manure of fertilizer should be encouraged, and this could be facilitated by increased cooperation between farmer's unions.

Mineral N is released slowly when solid organic fertilizers are used so N delivery can be better coupled with crop needs over time. This may decrease the need of synthetic fertilizers, thus saving energy and avoiding emissions produced beyond the boundaries of the farm during the industrial Haber-Bosch process of N fixation. In contrast, since the N content of manures is normally lower than that of synthetic fertilizers, the amount of organic matter to be applied in order to fulfil crop needs is high, so an increase in transport expenses and emissions would be expected unless manures are applied nearby the source.

Replacing synthetic fertilizers with organic ones is applicable to field crops such as cereals and oilseeds, given their high N demand. It is applicable to both irrigated and rain-fed systems, under Mediterranean conditions. Medium-textured and well-drained soils are the most suitable for this practice since they can counterbalance the N_2O denitrification losses associated to high C-content organic amendments (Velthof et al., 2003), whereas poorly-aerated soils tend to stimulate denitrification (Rochette, 2008). Technical issues related to temporal and spatial availability of animal manures must be considered. Intensive livestock production systems are often decoupled from agricultural systems. This causes mismatches between manure production and crop requirements, resulting in manure excess at a local scale. Thus, manure has to be transported to longer distances and/or treated before being applied, resulting in higher manure management costs (Teira-Esmatges and Flotats, 2003; Flotats et al., 2009).

The N_2O emission reduction at plot scale depends on the form of manure used. Solid manures have proved to significantly decrease N_2O emissions (ca. 23%) in Mediterranean systems (Aguilera et al., 2013b) and to have the potential to increase C sequestration in the long term (Ding et al., 2012). Webb et al. (2004) observed that solid manure incorporation decreased N_2O emissions, but Thorman et al. (2007a) found no consistent effect of incorporation of pig or cattle farmyard manure on such losses, except when denitrification is likely to be intense. However, as the readily-degradable C is mainly lost during the storage stage of solid manures, the C added to soil by incorporation will have less effect on the metabolism of denitrifiers (Webb et al., 2010). Overall, incorporation of solid manure in Mediterranean regions appears to reduce or have no impact on N_2O emissions (Table 1).

For liquid manures (i.e., slurries), no significant differences have been observed when these substitute synthetic N sources. This seems to be a consequence of the strong similarities between available N, in the form of NH_4^+, in both fertilizer types (Meijide et al., 2009; Plaza-Bonilla et al., 2014a). Other studies indicate that the method of slurry application is a key variable driving N_2O emissions from agricultural soils. According to a meta-analysis by Hou et al. (2015), injection of slurry could significantly increase direct emissions compared with broadcast application. However, in Mediterranean areas, dry matter content of slurries under dry weather conditions is normally high, thus reducing the potential of implementing injection practices. In cases of implementation, soil conditions appear to be the key factor affecting the direct and indirect N_2O emission pattern after slurry injection (VanderZaag et al., 2011). The addition of readily-mineralizable C from slurry has been shown to be the main driver for increasing emissions of N_2O by denitrifiers (Webb et al., 2010). If slurries are applied to crops, a social constraint related to smells and health issues may arise (Cole et al., 2000). This could be alleviated by restricting their use near towns or populated areas. Additionally, accumulation of heavy metals in the soil (e.g., zinc and copper present in animal diet) may represent a barrier for using these organic materials (Berenguer et al., 2008) (Table 2). There are also risks of antibiotic contamination of soils, and leaching when using manure (Díaz-Cruz et al., 2003).

C. Nitrification and urease inhibitors

Nitrification inhibitors (NIs) deactivate the enzyme responsible for the first step of nitrification, the oxidation of NH_4^+ to NO_2^-. By reducing nitrification rates, and subsequently the substrate for denitrification, the use of NIs may lead to reductions of N_2O emissions ranging from 30 to 50% (Huérfano et al., 2015) (Table 1).

Nitrification inhibitors are used in a wide range of agro-climatic regions (Akiyama et al., 2010; Gilsanz et al., 2016). In Mediterranean soils, NIs have shown high mitigation efficiency in rain-fed and irrigated fields, with a likely indirect effect on denitrification in the latter systems (Meijide et al., 2010). Soil texture may regulate mitigation efficiency (Barth et al., 2008) but to a limited extent, since soil texture has been shown to have a small influence on the inhibition of nitrification (Gilsanz et al., 2016)

Other soil parameters such as pH (with better performance in acidic soils) or organic C may affect the efficacy of NIs (Robinson et al., 2014; Marsden et al., 2015), especially for dicyandiamide (DCD), which explains the high efficiencies reported by studies performed in low-C Mediterranean soils. An inverse relationship between the inhibitory effect and temperature has also been described (Gilsanz et al., 2016), and should be considered when choosing the optimum application timing in each season.

The main limitation for implementation of NIs is the increase of fertilization costs (Timilsena et al., 2015). This could be counterbalanced by an increment in crop productivity (Abalos et al., 2014a). A potential enhancement in crop N use efficiency (Abalos et al., 2014a) may reduce N losses and may thus decrease the rate of synthetic N applied, reducing fertilization costs. Moreover, the use of inhibitors could simplify the task of fertilization by reducing the number of required applications, or by allowing for a greater flexibility in the timing of fertilizer application (Linzmeier et al., 2001).

Urease inhibitors (UIs) are used to reduce the activity of the urea hydrolase enzyme. Therefore, they can only be used when urea or urea-containing fertilizers (including organic sources) are used. Originally developed to reduce NH_3 volatilization, recent research has shown that these products may also reduce N_2O emissions (Sanz-Cobena et al., 2012; 2014a). Among the various types of UIs available, N-(n-butyl) thiophosphorictriamide (NBPT) has received the greatest commercial use (Sanz-Cobena et al., 2008; Abalos et al., 2014a). Recent studies have evaluated the effectiveness of NBPT to abate N_2O emissions in Mediterranean cropping systems, showing a high mitigation potential in an irrigated maize-field with nitrification-favoring conditions (55%; Sanz-Cobena et al., 2012), and in a rain-fed barley crop (86%; Abalos et al., 2012). An incubation experiment confirmed that the efficacy of the inhibitor to abate N_2O emissions is realized under conditions of low soil moisture (WFPS≤55%) (Sanz-Cobena et al., 2014a), common in Mediterranean semi-arid areas due to the scarce rainfall. The efficiency of UIs is expected to be highest in alkaline soils (frequent in Mediterranean climates), and is also generally higher in coarse-textured soils and at high N fertilization rates (Abalos et al., 2014a).

A cost-benefit analysis showed that mitigated N due to reductions in NH_3 volatilization when UIs are employed may serve to reduce fertilizer-N rates without incurring yield penalties (Sutton et al., 2015) (Table 1). The N rate reduction would decrease total fertilizer costs and partially offset the higher cost of urea treated with UIs. Further, reduced N rates may have additional environmental benefits such as reduction in NO_3^--leaching. However, such findings were obtained from studies in temperate climate, and remain to be confirmed under Mediterranean conditions.

2.1.2. Irrigation technology

Soil moisture, expressed as WFPS, is a key factor affecting N_2O losses (del Prado et al., 2006; García-Marco et al., 2014), hence the potential for N_2O mitigation linked to irrigation technologies is high (even above 50%) (e.g., Sánchez-Martín et al., 2010a, 2008; Guardia et al., 2016) (Table 1). The lower amounts of water applied in subsurface drip irrigation (SDI) or normal/superficial drip irrigation (DI) through more frequent irrigation events, generate “dry” and “wet” areas in the soil, lowering the overall soil moisture and favoring nitrification over denitrification (Sánchez-Martín et al., 2010a), thus reducing N_2O emissions (Table 1). Drip irrigation systems have shown an N_2O EF of only 0.18%, compared to an EF of 1 % in sprinkler systems (SI), showing the mitigation potential of irrigation technologies in the Mediterranean region (Cayuela et al., this issue).

Optimized irrigation techniques to decrease GHGs emissions on Mediterranean regions are particularly used in perennial crops and intensive vegetable cropping systems (SDI, DI), and in paddy soils (water table management).

Subsurface drip irrigation has been shown to be beneficial in terms of increased yield, improved crop quality, and reduced agronomic costs (e.g., for weed control or water applied) (Ayars et al., 2015), but there are some technical and economic constraints associated with conversion, automation and maintenance. Indeed, the use of different irrigation systems results in distinct water use patterns. This is particularly important in Mediterranean systems, where irrigation needs to be optimized, due to limited water resources during summer crop growth periods. The most efficient irrigation system from the water use perspective is subsurface drip irrigation (SDI), followed by normal/superficial drip irrigation (DI) and sprinkler (SI). In contrast, whereas furrow irrigation (FI) results in the highest water consumption rates, thus coincident with N_2O mitigation technology.

2.1.3. Fertigation

Irrigation combined with split application of N fertilizer dissolved in the irrigation water (i.e., fertigation) is ideally suited for controlling the placement, time and rate of fertilizer N application, thereby increasing N use efficiency. This fertilization strategy is highly relevant in a context of increasing drought periods due to climate change in Mediterranean agro-ecosystems (Abalos et al., 2014b). Reductions in direct N_2O emissions between 30 and 50% compared with traditional fertilization and irrigation practices have been reported for Mediterranean fertigated crops, mostly due to an effect on nitrification rates (Kallenbach et al., 2010; Schellenberg et al., 2012; Kennedy et al., 2013; Abalos et al., 2014b; Vallejo et al., 2014) (Table 1). Since this is a relatively new methodology, there could be initial economic barriers associated with conversion from furrow or sprinkler (Table 2). Technical and economic barriers associated with maintenance may also exist; a problem that automation may partially overcome, easing irrigation and fertilization activities (Thomson et al., 2000). Conversely, fertigation may serve to reduce costs due to input savings (e.g., water, fertilizers) and increases in crop quality and productivity (Kennedy et al., 2013; Ayars et al., 2015).

2.2. Agronomic practices affecting CH_4 emissions

Mediterranean agricultural soils produce large CH_4 emissions in flooded crops (e.g. rice) through methanogenesis, representing 6% of all CH_4 production from agricultural sources. Water table management has been proven to significantly reduce CH_4 losses in non-Mediterranean climates (Yagi et al., 1997; Kudo et al., 2014; Liang et al., 2016). By decreasing the flooding period, both methanogenesis and CH_4 evasion through the water table, one of the CH_4 transport pathways, are limited. This leads to lower emissions and reduces water consumption, a crucial goal to improve the sustainability of Mediterranean agro-ecosystems (Rizzo et al., 2013; 2015) (Table 1).

Methane emissions also depend on the incorporation of organic matter (mainly crop residues). Increases in CH_4 emissions from rice production were reported when straw was added from 0 up to 7 t N ha^{-1} in a Mediterranean cropping system (CH_4 emission ranging from c. 100 to c. 500 kg CH_4 ha^{-1} yr^{-1}; Sanchis et al., 2012) (Table 1).

With regard to rice straw management strategies, recommended practices for enhancing GHG mitigation are composting rice straw, straw burning under controlled conditions, recollecting rice straw for biochar production, generation of energy, using it as a substrate, or source of other by-products with added value.

In non-flooded Mediterranean systems, the effect of fertilizer application rate on soil CH_4 uptake has been found to be positive (Meijide et al., 2016), negative (Guardia et al., 2016) or neutral (Plaza Bonilla et al., 2014). Variable effects, depending on organic or synthetic fertilizers on CH_4 sink capacity, were reported by Sánchez-Martín et al. (2010b). The lower CH_4 uptake following the application of high C-content amendments has been related to changes in soil porosity and enhancement of soil respiration rates, promoting anaerobic microsites and consequently reducing methanotrophy (Le Mer and Roger, 2001).

2.3. Agronomic practices affecting C sequestration

Levels of OM in Mediterranean soils are generally low and are expected to decrease further in many Mediterranean areas in the coming years (Davidson and Janssens, 2006) as a result of generalized low C inputs and increased soil organic carbon (SOC) decomposition rates associated with rising temperatures (e.g., Al-Adamat et al., 2007) thus increasing the GWP of Mediterranean agro-ecosystems.

Management practices aimed at increasing SOC stocks must target a positive balance between C inputs and outputs through the reduction of SOC losses (Plaza-Bonilla et al., 2016), the increase of organic C inputs into the soil, or both (Aguilera et al., 2013a; Six et al., 2004). Most practices leading to increasing SOC content include reduced soil tillage, careful management of crop residues and agroindustry products in herbaceous crops, and cover cropping in orchards. These practices have relevant co-benefits through improved soil physical, chemical and biological quality (Lal, 2011; Lassaletta and Aguilera, 2015), enhanced crop productivity, reduced dependence on external inputs (Smith and Olesen, 2010) and lower soil erosion rates.

2.3.1. Reduced soil tillage

Reduction or complete cessation of tillage decreases the direct incorporation of fresh organic debris into deeper soil layers. The absence of tillage (NT) slows down aggregate turnover and, in turn, increases the physical stabilization of SOC within soil aggregates (Álvaro-Fuentes et al., 2008; Plaza-Bonilla et al., 2010). An approximate annual increase of 1% in SOC when tillage is avoided in Mediterranean croplands has been observed (own estimation from Aguilera et al. 2013a). This is above the 0.4% targets of recent initiatives for sustainable soil conservation (http://4p1000.org). The response under reduced tillage (RT) was variable and of similar magnitude, with average accrual rates of 0.32-0.47 Mg C ha^{-1} yr^{-1} compared to conventional tillage management (CT) (Sánchez et al., 2016). In semiarid conditions (400 mm of total rainfall), Guardia et al. (2016) indicated that NT fixed 0.5 Mg C ha^{-1} yr^{-1}, whereas 0.06 Mg C ha^{-1} yr^{-1} was accumulated in the soil under RT practices. Estimates are highly dependent on the soil depth used for the calculation, since vertical SOC distribution in NT and CT systems is different (Cantero-Martínez et al., 2007). Further, the assumption of a steady and linear C sequestration may not hold true, because the annual C accumulation rate tends to decrease in the long-term (Álvaro-Fuentes et al., 2014).

No-tillage practices are more commonly used in rain-fed systems; but they are also suitable for irrigated (although RT is more recommended in these systems), extensive, intensive and organic systems with well-drained soils. In water-limited regions, such as dryland Mediterranean areas, NT enhances soil water retention potential, and has a positive effect on biomass production and crop residue inputs (Lampurlanés et al., 2016). The greater soil water retention potential under NT is the result of reduced evaporation due to the mulch protection, and enhanced soil water infiltration due to the higher structural stability at the soil surface.

The reduction or cessation of tillage requires specific management according to the climatic zone (SmartSOIL, 2015). Reduced tillage is an accepted practice by an increasing proportion of farmers, although initial investment cost for specific seeding machinery can constrain farmers' willingness to adopt RT or NT. It usually leads to net cost reductions, despite the initial investment (Sánchez-Girón et al., 2004) since farmers save labor time and fuel inputs compared to conventional tillage (Álvaro-Fuentes et al., 2014; Sánchez et al., 2014, 2016; Guardia et al. 2016; SmartSOIL, 2015). Even so, NT practices need to be accompanied by the application of herbicides, which may increase costs and produce pollution in soil and water bodies if improperly managed (Annett et al. 2014). Efforts are being made to promote NT practices with decreased use of phytochemicals (e.g. Sans et al. 2011; Armengot et al., 2014).

2.3.2. Crop rotations and cover crops

Long crop rotations have been proposed in rain-fed Mediterranean cropping systems to enhance C sequestration and restore soil fertility and structure (Benlhabib et al., 2014). The effect of crop rotations on C sequestration is highly dependent on time with no significant effect reported in short-term studies (López-Bellido et al., 1997; Hernanz et al., 2002; Martin-Rueda et al., 2007). However, positive effects in long-term experiments (>15 years) could appear if crop biomass is properly managed after harvest

(Masri and Ryan, 2006; López-Bellido et al., 2010; Martiniello and Teixeira da Silva, 2011). For instance, a wheat-chickpea crop rotation under CT, showed a C sequestration rate of 0.53 Mg C ha^{-1} y^{-1} during a 20-year period, compared with wheat monoculture (López-Bellido et al., 2010). The effect of crop rotations on SOC stocks is also dependent on the type of crops included in the rotation (Triberti et al., 2016) and the management of crop residue. The introduction of perennial crops to rotations has shown benefits for SOC stock and soil quality (Di Bene et al., 2011; Pellegrino et al., 2011). The substitution of bare fallows by any crop (usually used to improve water and nutrient availability for the following crop) has been associated with SOC stabilization in NT systems (Álvaro-Fuentes et al., 2009), and to reduced soil erosion (Boellstorff and Benito, 2005). The effect on C sequestration of the inclusion of grain legumes in rain-fed yearly rotations is dubious, due to their low biomass production, although their conversion to stabilized soil organic matter could be more efficient than that of cereals (Carranca et al., 2009). Consequently, the highest potential of fallow and legumes for mitigating GHG from these types of cropping systems comes from the avoidance of fertilizer production emissions.

Implementing crop rotations requires more detailed planning compared to monocultures (e.g., selecting crop species/sequences and nutrient and weed control practices), which can constitute a management constraint. On the other hand, reduction of fertilizer, pesticide and herbicide needs, and possible crop yield and soil quality improvements in the long term, added to the low investment and operational costs to implement the practice, may encourage farmers to establish this traditional crop management practice (Ferrio et al., 2007). Moreover, some legume species and cultivars (e.g., green beans, peas, etc.) can represent high-value crops, particularly in vegetable crop rotations. In forage cropping systems, leguminous species can improve the forage quality and therefore the economic profit (Rochon et al., 2004; Kalac, 2011). Crop rotations (particularly those which involve legumes) are included in the greening requirements of the European Union Common Agricultural Policy (EU CAP) incentives (crop diversification), thus encouraging implementation among farmers (Ingram et al., 2014).

Cover cropping (CC) in herbaceous cropping systems involves the use of catch crops or green manures during the intercrop period of irrigated cropping systems (intra-annual rotation) or substituting bare fallows in rain-fed cropping systems (inter-annual rotation). In fruit orchards, CC involves the use of understory vegetation between tree rows or in the whole soil surface. Catch crops are intended to reduce nutrient losses in soils that are prone to greater N leaching losses (e.g. sandy or highly fertilized soils). In terms of C sequestration, the use of CC has been proposed as a mean to enhance SOM and labile C pools by incorporating plant material into the soil (Veenstra et al., 2007). Average C sequestration potential of winter CCs (cultivated in the intercrop period of summer crops) has been reported at 0.32±0.08 Mg C ha^{-1} yr^{-1} at the global level (Poeplau and Don, 2015). For Mediterranean areas, González-Sánchez et al. (2012) studied cover crops in woody cropping systems of Spain, reporting average C sequestration rates of 1.54 and 0.35 Mg C ha^{-1} yr^{-1} in studies of less and more than 10 years, respectively, while Aguilera et al. (2013a) calculated an average carbon sequestration rate of 0.27 Mg C ha^{-1} yr^{-1} for all types of cover crops in a meta-analysis of Mediterranean cropping systems.

Application of CCs is limited during seasons with water scarcity. General lack of knowledge of the best CC management practices for optimizing both environmental and economic profits limits the correct implementation of CC. Selection of plant species, the management of residues and the kill date are crucial factors (Gabriel et al., 2012;

Alonso-Ayuso et al., 2014; Sanz-Cobena et al., 2014b) likely influencing the yields and N uptake efficiency of the succeeding cash crop (Míguez and Bollero, 2005, Tonitto et al., 2006). Reduction of fertilizers required for the subsequent crop, (especially when grain legumes are used as green manure), and gain of secondary products (e.g., animal feed) can deliver positive economic benefits (Gabriel et al., 2013; Scherback et al., 2014); usually outcompeting sowing and killing costs (Table 2). Furthermore, CCs prevent soil erosion, runoff and sediment losses (Hargrove, 1991; Blanco-Canqui et al., 2015), improve soil structure, N supply and water retention capacity (Quemada and Cabrera, 2002; Suddick et al., 2010), reduce leaching (Bugg et al., 2007), improve soil microbial quality (Balota et al., 2014) and reduce soil salinity during the early stages of the cash crop (Gabriel et al., 2012).

2.3.3. Management of crop residues and agroindustry by-products

Estimating the GHG mitigation potential of using crop residues and organic by-products from agroindustry in Mediterranean areas implies accounting the net GHG balance when they are used as: (i) soil amendments to improve SOM and enhance SOC sequestration (Aguilera et al., 2013a), (ii) feedstock for bioenergy production (e.g. Di Giacomo and Taglieri, 2009; Spinelli and Picchi, 2010), (iii) co-substrate for composting (e.g. Santos et al., 2016) , (iv) feed for livestock (e.g. Molina-Alcaide and Yáñez-Ruiz, 2008) or (v) construction materials (e.g. animal beds, buildings). Also, we must have a realistic estimate of the current fate of these organic matter streams and the sustainability or economic issues (Pardo et al., 2013) that may jeopardize the realization of such potential. To our knowledge no such study has been made for the whole Mediterranean area. Pardo et al. (this issue), estimated for the Mediterranean coastal areas in Spain reductions of 4.3 Tg CO_2eq yr^{-1} (about 11% of total agricultural emissions in Spain in 2014) if available local by-products from agri-food industries was codigested with existing manure and applied to the nearby available agricultural soils. This study suggests that, despite the overall large stocking of crop-residues and by-products in the Mediterranean basin (FAOStat, 2016), the potential for their use in cropping systems may be reduced by its availability nearby.

The potential to increase SOC levels by using agroindustry by-products, as in crop residues, depends on their composition and degradability. However, agroindustry by-products vary widely in their chemical composition and therefore in their degradation rates. For example, olive and mill waste as they have very low degradation rate in the soil have been found to be good amendments to increase SOC when applied to the soil (Saviozzi et al., 2001; Sanchez-Monedero et al., 2008).

Besides the potential direct GHG reduction that any strategy involving the return of the crop residues and agroindustry by-products to the soil may cause, (e.g., Kassam et al., 2012, Gonzalez-Sanchez et al., 2012; Aguilera et al., 2013a; Plaza-Bonilla et al., 2015) applying these materials, treated or un-treated, as soil amendments can also deliver environmental co-benefits, such as erosion reduction when they as raw are used for mulching (Blavet et al., 2009; Jordán et al., 2010) or, in general, allowing closing the nutrient cycles, with associated potential reductions of fertilizer use and reductions in the draught force and fuel consumption for soil tillage (Peltre et al., 2015). Trade-offs, however, may occur with some of the strategies that may results in larger GHG mitigation potential. For example, the use of crop residues on the soil surface might pose a risk of fire in some Mediterranean areas (Luna et al., 2012) and, sanitary, pollution and legal constraints may apply, especially if the by-product is applied to crops e.g. fresh vegetables without pre-treatment (Table 2).

Composting and anaerobic digestion of agroindustry by-products are common treatments that can improve the properties of the organic matter and can also provide additional overall GHG reductions (del Prado et al., 2013). The composting process has relatively low associated GHG emissions (Pardo et al., 2015) and can lead to moderate to high SOC sequestration rates when used as soil amendments (Aguilera et al., 2013a). Long term humic-clay associations promote a more efficient protection of SOM and long-lasting C sequestration in amended soils. The composted material will lower the soil pH, reducing the decarbonation process in soils developed over calcareous materials (common in the Mediterranean basin). Anaerobic digestion of agro-industry by-products reduces overall GHG emissions through the generation of biogas. The conversion of OM into biogas (i.e., CO_2 and CH_4) involves a fraction of C that is released to the atmosphere, instead of being applied to the land. Therefore, although digestate application increases soil C storage and produces benefits over soil quality in the long term, the potential for C sequestration (per unit of initial residue amount) could be lower when compared with undigested materials. On the other hand, the high nutrient availability of anaerobically digested organic wastes makes digestate an economically viable substitute of mineral fertilizer (Arthurson, 2009).

Sewage sludge is currently applied to agroecosystems, especially to degraded soils of Mediterranean areas (Albiach et al., 2001; Fernández et al., 2009) due to its high OM. However, the labile OM forms present in sewage sludge and the high amounts usually applied (Franco-Otero et al., 2011) may increase CO_2 emissions due to increased soil respiration (Flavel et al., 2005; Song and Lee, 2010). Sludge use is highly constrained by fresh water pollution and availability issues. It is likely that increasing social and political environmental concerns, reflected in national and international normative, will further extend the use of wastewater treatment systems thus increasing sludge production. In this context, the EU Landfill Directive 99/31/EC (CEC, 1999) banning the landfilling of sewage sludge (Klee et al., 2004) should lead to a better reuse of sewage sludge, thus reconnecting urban and rural environments and ensuring the absence of risks for both the society and the environment. In this sense, further studies are needed to assess the impact of sewage sludge on (e.g.) antibiotic resistance in soil microbiota (Chen et al., 2016).

Biochar (a solid by-product generated by pyrolysis) application to soils has been suggested as a means of reducing atmospheric CO_2 concentration. Biochar's climate change-mitigation potential relies on its highly recalcitrant nature, which decreases the rate at which vegetation C is released to the atmosphere (Woolf et al., 2010). Biochar's mitigation potential depends on production process, and further experimental assessments of its efficiency under Mediterranean conditions are required (Hussain et al., 2016).

3. Side-effects associated to selected GHG mitigation practices

3.1. GHG emissions

Specific management practices primarily target the mitigation of a single GHG (e.g. decreased soil tillage aimed at increased soil CO_2 sequestration) may promote the release (trade-off) or mitigation (win-win) of other GHGs (e.g. N_2O or CH_4).

Enhanced direct N_2O emissions have been observed after NT in the short-term (Six et al., 2004), especially in poorly drained soils (Rochette et al., 2008). On the long term, increased soil porosity in NT systems, counterbalances the greater WFPS levels typically found in NT compared to tilled soils (Plaza-Bonilla et al., 2013; van Kessel et al., 2013). Conversely, NT can reduce indirect N_2O emissions due to lower runoff and N leaching (Holland, 2004; Soane et al., 2012).

In the case of crop rotations with bare fallow (BF), Sánchez-Martín et al. (2010b) showed negative N_2O fluxes in a fallow period between two irrigated onion crops under Mediterranean conditions. Under similar climatic conditions but in a rain-fed crop, Téllez-Rio et al. (2015) observed lower N_2O emissions from a wheat crop preceded by a fallow period than from a monocrop of the same cereal.

For crop rotations including CCs, the effect on N_2O emissions needs to be assessed by differentiating the intercrop and the cash crop periods. During the intercrop, contrasting results have been obtained. The meta-analysis of Basche et al. (2014) pointed out an overall enhancement of N_2O losses, particularly in the case of legume-CCs. These results were supported by Guardia et al. (2016) in a field experiment in Mediterranean conditions. Conversely, during the subsequent cash crop period, CCs as opposed to BF have potential to decrease N_2O emissions due to the lower requirement of N fertilizers. The same authors showed that synthetic N applied to a maize crop preceded by vetch (a legume) could be decreased by 25% without yield penalties. However, also under Mediterranean conditions, neither Sanz-Cobena et al. (2014b) nor Guardia et al. (2016) observed a significant effect of catch crop management on N_2O emissions when considering the whole crop and intercrop cycles. Since the effect of CCs on direct N_2O losses is negligible (particularly when considering the whole cropping cycle and integrated fertilization management) CCs mainly reduce indirect N_2O emissions associated with N leaching (Gabriel et al., 2012; Quemada et al., 2013). In any case, both BF and the use of legumes in yearly rotations decrease the GHG emissions from N fertilizer manufacturing, making crop operations (e.g., machinery, agrochemicals manufacturing, etc.) the main source of GHG emissions in these systems (Aguilera et al., 2015a; Guardia et al., 2016).

Biochar has attracted attention as a strategy for mitigating N_2O emissions from agricultural soils, along with the initial concept of increasing SOC stocks. Biochar was found to decrease N_2O emissions by close to 50% (Cayuela et al., 2015), with soils from Mediterranean origin showing variable but large mitigation potential, up to 90% according to lab studies of wood biochar (Cayuela et al., 2013) . However, field studies under Mediterranean conditions have shown small to no significant reductions (Castaldi et al., 2011; Suddick and Six, 2013; Pereira et al., 2015), or even a slight increase in N_2O emissions (Sánchez-García et al., 2016). These different outputs between lab and field studies were probably due to the fact that laboratory conditions were not finally reflected on the field (Cayuela et al., 2014), and suggests that further experiments using a range of soil types, crops (absence of perennial and horticultural crops) and management practices is required. The effectiveness of biochar to significantly decrease N_2O emissions depends on the soil type (Sánchez-García et al., 2014), the N fertilizer used (Nelissen et al., 2014) and, ultimately, on the main pathways leading to N_2O formation (nitrification *vs.* denitrification). Biochar from woody materials (low C/N ratios) produced by slow pyrolysis at high temperatures (>500 °C; molar H:Corg<0.3) have shown the highest mitigation potential (Cayuela et al., 2014; 2015).

3. 2. Non-GHG emissions

Ammonia volatilization, nitric oxide (NO) and NO_3^- leaching are the main pathways of non-GHG pollutant release to the environment from Mediterranean agricultural soils. Whereas NO contributes to the formation of ozone, and influences air quality, both NH_3 and NO_3^- losses also indirectly affect emissions of N_2O (IPCC, 2006).

In rain-fed systems, NO_3^- leaching normally occurs in autumn and is mostly driven by episodic precipitation events and external N inputs. In irrigated systems, N losses through leaching occur in summer due to high irrigation and N fertilization rates. Ammonia emissions are common in both rain-fed and irrigated cropping soils if urea and ammonium-based fertilized are applied to the soil surface.

Adjusting N fertilization rates to crop needs may have positive side-effects on the abatement of both NH_3 volatilization and NO_3^- leaching (Quemada et al., 2013; Sanz-Cobena et al., 2014c). The use of solid manure can lower N losses through reduced leaching (Sanchez-Martín et al., 2010) and N_2O emissions (Meijide et al., 2007; 2009), due to enhanced microbial and plant immobilization of N (Table 1). The application of liquid manure (slurries) can improve soil structure, decreasing the risk of N leaching in the medium term (Zavattaro et al., 2012; Plaza-Bonilla et al., 2013), although it can increase it in the short term when applied at high rates (Yagüe and Quilez, 2015).

Manure application in the field can trigger NH_3 volatilization (e.g. Sanz et al., 2010; Vinguria et al., 2015) if no NH_3-abatement strategies are applied (Sanz-Cobena et al., 2014c). Slurry injection technologies have been shown to reduce NH_3 emission by 40-90% compared with broadcast application (Webb et al., 2010). However, this may leave more mineral N available to be lost in the form of e.g., NO_3^- and N_2O if soil conditions favor denitrification (high WFPS) (Sanz-Cobena et al., 2014c). On well-drained arable soils, injection can reduce N losses, as it reduces NH_3 volatilization while it has little effect on N_2O emission rates. In Mediterranean agriculture, slurry injection is still a marginal practice, but may have a great potential for NH_3 abatement without compromising N_2O mitigation due to dry soil conditions that are unfavorable for denitrification. Immediate incorporation of manure (pig, cattle and poultry manure) into the soil by ploughing may reduce up to 90% of NH_3 compared with no ploughing (Webb et al., 2010). Ammonia abatement will decrease to 50% if soil incorporation is delayed for some hours (Dell et al., 2011), or incorporation systems other than ploughing are used (e.g., discs, tines; Thompson and Meisinger, 2002). Fertilizer injections and tilling within the first 24 h after application are not popular among farmers because of the additional costs and technical difficulties associated.

In the case of digested agroindustry by-products, the increase in NH_4^+ associated with the transformation process improves fertilizer potential, but may also enhance N emissions through NH_3 volatilization. The final effect on direct and indirect NO emissions will be determined by the complex interactions involved in the soil-plant system, which are influenced by the composition of the organic amendment, but are tightly controlled by the soil conditions (e.g. water content, temperature) and the time and method of application (Thorman et al., 2007). Similarly, for sewage sludge applied to croplands as soil amendments, large amounts of N in NH_4^+ form may be released, providing a substrate for nitrification (Kleber et al., 2000) and thus increasing NO emissions (Roelle and Aneja, 2002). Techniques to stabilize sludge improve the soil retention of organic C (Dere and Stehouwer, 2011) and reduce the risk of N leaching (Correa et al., 2006) due to the low proportion of available N (15-20%). In contrast, thermal-drying of sludge causes an increase in easily-mineralizable organic N (Tarrason et al., 2008), with readily plant-available NH_4^+ of up to 85% (Gendebien et al., 2008). This may lead to, not only to higher GHG emissions due to enhanced activity of nitrifiers and denitrifiers, increasing risk of NH_3 volatilization.

The inclusion of a NI with any NH_4^+-based N fertilizer will retain N in the soil in the form of NH_4^+, thus reducing potential losses by NO_3^- leaching (Quemada et al., 2013). By inhibiting nitrification, NIs can also mitigate NO emissions (Qiao et al., 2015; Guardia et al., 2016). The expected increase on NH_4^+ in the upper soil associated with the use of NIs may increase the risk of NH_3 volatilization if environmental and weather conditions are favorable for this process and the fertilizer is applied to the soil surface. On the other hand, the production and transport of inhibitors may increase emissions of CO_2. Reductions in NH_3 volatilization induced by UIs may increase soil mineral N prone to be lost as NO_3^- leaching, which would eventually increase indirect N_2O emissions. However, the few field investigations carried out under Mediterranean conditions have not shown any significant UI effect on N leaching (e.g., Sanz-Cobena et al., 2012).

Improved irrigation techniques have been shown to reduce NO_3^- leaching (Quemada et al., 2013) as a result of lower soil moisture and a lower proportion of wet soil surface, but may potentially increase NO emissions (Sánchez-Martín et al., 2010a). In the case of fertigated soils, indirect N_2O emissions from leached NO_3^- would be reduced due to lower irrigation rates and higher irrigation frequency (avoiding deep percolation), as well as better synchronization between N supply and plant N needs (Quemada et al., 2013).

3. 3. Crop yields

Adjusting N fertilization rates to crop needs, if properly done, does not have negative effects on crop yields (Yagüe and Quilez, 2010), but only on reduced N losses. Similarly, the use of organic amendments instead of synthetic fertilizers does not have a negative effect on crop yields *per se*. As occurring with synthetic fertilizers, applying solid manures only as N fertilizer could decrease yields if N application rates (and/or timing) are not precisely adjusted to crop requirements (Abalos et al., 2013) (Table 1).

This may result in an effective mitigation per surface area but the yield-scaled N_2O emissions could increase.

Solid manures are usually applied in combination with synthetic fertilizers or liquid manures to achieve adequate N application rates. For slurries, increases in cereal yields have been reported, presumably due to a more balanced nutrition (Plaza-Bonilla et al., 2014a). However, in more productive areas (e.g. irrigated or sub-humid) and high-yielding crops (e.g. maize) farmers tend to complement slurry application with synthetic fertilizer as a top-dressing application (Bosch-Serra et al., 2015). The use of urease or nitrification inhibitors in combination with synthetic fertilizers has shown slightly positive or negligible effects on crop yields (Abalos et al., 2013). No significant effect of 3, 4-dimethyl pyrazole phosphate (DMPP) on crop yields has been measured, while increases in yields (5-10%) have been measured when using DCD (Vallejo et al., 2005; Abalos et al., 2014a; Huérfano et al., 2015).

Diversified crop rotations have shown to improve yields (Lopez-Bellido et al., 2000; López-Fando and Almendros, 1995) (Table 1). On the contrary, the presence of BF in rotations is usually associated with decreased SOC contents (Álvaro-Fuentes et al., 2008; Ryan et al., 2009) enhancing the cropping system GWP, but also affecting soil fertility and the yield-scaled GHG budget. The benefits on crop yield and direct N_2O emissions (considering the whole intercrop-cash crop cycle) are enhanced when using legume CCs (Quemada et al., 2013; Doltra and Olesen, 2013; Tonitto et al., 2006) but there may be drawbacks for direct mitigation of N_2O emissions during the intercrop period (Basche et al., 2014; Guardia et al., 2016), as well as for preventing N leaching. Further research should analyze these trade-offs in the short- and long-term, considering both direct and indirect N_2O and other GHG emissions.

In the case of soil management practices, although highly dependent on pedoclimatic conditions, increases up to 20% in yields have been reported in Mediterranean environments under reduced tillage (Cantero-Martínez et al., 2007; Pittelkow et al., 2015a) with some exceptions (Pittelkow et al., 2015b).

4. Effect of agricultural practices on the total GHG budget of rain-fed and irrigated cropping systems

The main management practices affecting C sequestration, N_2O and CH_4 emissions have been discussed, so the most promising measures can be selected, considering the overall GHG balance in each specific Mediterranean agro-ecosystem (Table 3). The dominant GHG sources of each cropping system and each particular area (local pedoclimatic conditions) should be considered for prioritizing the adoption of efficient techniques, but also taking into account all practices that could provide an optimum balance between GHG mitigation and crop yields while saving/maintaining farm expenses or leading to an efficient use of available resources.

The study of Aguilera et al. (2015a) pointed out that the main GHG sources in herbaceous cropping systems in Mediterranean areas were emissions from machinery due to the low direct GHG emissions in these systems. Guardia et al. (2016), in a non-irrigated cereal-legume rotation, also confirmed that the relative weight of N_2O losses was lower than that of farm inputs and operations, while C sequestration was the main GHG component under NT adoption. Despite some uncertainties and variability that could be attributed to the C sink (e.g., the depth considered for calculation, the decrease of annual sequestration rate in the long term) (Álvaro-Fuentes et al., 2014), it appears that practices such as NT/RT combined with crop rotations including legumes and cover crops, without removal of crop residues, are the most promising for minimizing fuel consumption and external inputs (e.g. conservation agriculture practices, as conventional ones, might rely on the use of pesticides), and promote C sequestration (Table 3). These practices may provide the best GHG balance in rain-fed Mediterranean herbaceous crops, without negative side-effects on crop yields or N losses. Adjusting N rates to crop needs may improve the GHG balance of rain-fed herbaceous cropping systems through two components (N_2O emissions and CO_2 equivalents from production and transport of fertilizers) while reducing costs, so this practice should be encouraged in Mediterranean areas.

In summer irrigated crops, high N_2O losses can occur (Aguilera et al., 2013b). Consequently, agricultural practices based on an improved management of irrigation water (e.g., drip irrigation), N fertilization (e.g., adjusting N rates and timing, use of nitrification inhibitors) and both (e.g., fertigation) are the most promising measures in these agro-ecosystems. Since fruit orchards are broadly characterized by efficient water and fertilizer use (e.g., drip irrigation and drip-fertigation), other promising techniques are cover cropping (thus minimizing fuel consumption) and pruning-residue management for enhancing C stocks (Aguilera et al., 2015b) (Table 2).

Methane emissions are the main component of the GHG budget of paddy fields (Aguilera et al., 2015a), so mitigation efforts should focus on water management for minimizing these losses (see section 2.1.). Reducing water consumption in vegetable cropping systems may lead to substantial GHG emission reductions (Aguilera et al., 2015a).

5. Socioeconomic performance of agronomic measures and constraints to implementation

The degree of implementation of agronomic strategies proposed in this review differs among countries under Mediterranean climatic conditions. Even so, management strategies based on farmers' practices (e.g. crop rotations, cover cropping, etc.) are widespread, but there is room for increasing their application.

Adoption of conservation agriculture (CA) practices (i.e. coincidence in time and space of i) reduced tillage, ≤25%, or no-tillage; ii) >30% of soil cover, with mulch materials or living crops including CCs; and iii) crop rotations or associations) (FAO, 2011) in dry Mediterranean cropping systems has been reported by Kassam et al. (2012). According to this study, CA practices are implemented in 72 million ha (14% of the total cropland with this climatic regime). Outside the Mediterranean basin, where adoption of CA practices is still modest (c. average of 3% over total arable land) (Lahmar, 2010; Kassam et al., 2012), there are several countries and regions showing successful adoption of CA. These include the USA (16% of total cropland under no-tillage) (Kassam et al., 2012), central Chile (30% of rain-fed systems growth under CA practices) (Derpsch and Friedrich, 2009), South Africa and south Western Australia (CA adopted by 90% of farmers) (Llewellyn et al., 2009). In Mediterranean Europe, Spain is the country with the largest cropping surface under CA (650,000 ha, 5% of cropland, and 1,218,726 ha of perennial trees - mostly olives and grapes - in combination with CCs) (MERMA, 2010; González-Sánchez et al., 2015). In North and South African areas under Mediterranean conditions, the implementation of CA is, to date, sparse (Derpsch and Friedrich, 2009; FAO, 2011). Even so, cereal-based CA systems of Mediterranean regions of northern Africa and Southern EU (i.e. organic farming systems) frequently show coexistence of livestock (e.g. small ruminants) and cropping systems (e.g. olives), which facilitates CA practices such as crop rotations as well as the reusing of manures as fertilizers (Kassam et al., 2012).

Mitigation through water management approaches also presents a high potential. Spate irrigation dominates African regions under Mediterranean conditions (FAO, 2012). Irrigated crops are grown under full controlled irrigation, which includes surface, sprinkler and drip irrigation in the EU, EEUU and Oceania. Among the irrigation technologies used in Mediterranean cropping systems, furrows are still widespread in summer-irrigated crops, followed by increasing sprinkler irrigation systems (MAGRAMA, 2014). Surface irrigation with furrows was applied in 62% and 71% of the total irrigated cropland (14,249 ha and 3,297 ha) for maize and wheat, respectively, according to a survey based report focusing on farmers practices of the Ebro watershed (Spain) (Sisquella et al., 2004). Water-saving irrigation systems such as drip irrigation (both surface and subsurface) are still being developed (Zalidis et al., 2013).

Fertigation use is increasing, particularly in high-value crops (e.g. horticulture, orchards) which are very representative in Mediterranean areas. According to FAO (2014), around 9 million ha of cropland are currently under fertigation.

Nitrogen over-fertilization has been noticed in agricultural systems of high income economies, mostly in irrigated cropping systems. On average, 57% of the N crop uptake is over applied in Europe (Sánchez et al., 2016). This percentage is even higher in certain Mediterranean EU countries, such as Italy and Spain, where there are hotspots of intensive livestock production, leading to large quantities of manures normally surface-applied to croplands (Sanz-Cobena et al., 2014b). As an example, in maize crops of Catalonia and Aragón (NE Spain) farmers apply more than 400 kg N ha^{-1} in 84% of the cropping area due to application of both manures and synthetic fertilizers (Sisquella et al., 2004). According to expert judgement, this can be extrapolated to cropping areas of California (USA), Australia and Chile although, in these regions, surface application of manure is common on pasture and silage fields and some rangelands.

The implementation of technological mitigation solutions focusing on fertilization, such as urease and nitrification inhibitors, is expected to be limited in Mediterranean cropping systems, mostly due to associated extra costs for farmers. According to producers, the use of inhibitors increases cost of synthetic N fertilizer by 20% (Sutton et al., 2015). According to this, a larger expansion would be expected in high income economies (e.g. EU, EEUU and Australia), where there could exist subsidies for farmers to adopt this kind of technology.

Based on this analysis, there is large potential for implementing the strategies presented in this review. However, there are certain constraints that may make their implementation more difficult in the coming years.

5. 1. Constraints to management practice change

Constraints to management practice change by farmers, and the overall impact of these constraints on implementation of the practice, assessed by expert judgment are summarized in Table 2. The application of most of these agronomic measures can be hindered by economic constraints. Several practices require an initial investment for the acquisition of specific equipment (improved irrigation technology, fertigation, crop residues and agro-industry by-product management, low/no tillage). Economic constraints could also arise in the form of a regular cost due to possible yield penalties (N fertilization adjustment, organic fertilization, low/no tillage and cover crops). In the case of crop rotations and the use of crop residues and agro-industry by-products, these practices can reduce benefits from other economic activities.

Most agronomic measures described in this review are also accompanied by some kind of technical constraint. This mainly relates to N fertilizer adjustment, the substitution of synthetic fertilizer by manures, the application of sewage sludge, no/low tillage practices, cover crops and crop rotations. Some of these practices require additional work, such as soil sampling, or learning how to use or maintain new equipment (e.g. incorporation of manures, improved irrigation technology, fertigation and low/no tillage). Finally, low/no tillage practices can increase weeds and soil compaction problems, thus increasing the need for additional management practices, particularly the first years after adoption (Soane et al., 2012; Armengot et al., 2015).

Social constraints to management change are largely associated with farmer perceptions (Sánchez-Girón et al., 2004; Ingram et al., 2014; Sánchez et al., 2014; 2016). Conventional farmers can be reluctant to implement some of the practices because of strong traditions (e.g., crop residue management, no-tillage, cover crops) or having a perception of decreased productivity due to practice implementation (e.g., adjusting fertilization rates or shifting from synthetic fertilizers to manure). Further, new recommended practices (e.g., nitrification and urease inhibitors, biochar), which are not yet widespread among neighboring farmers can be negatively perceived (Hussain et al., 2016). A lack of training for practices with high technical or maintenance requirements (e.g., irrigation technology, cover crops, crop rotations, adjusting fertilization rates) may lead to management difficulties, or the misuse and decline of yields, in turn encouraging the negative perception of the practice's effectiveness (Cantero-Martínez et al., 2007; Abalos et al., 2013). Legal restrictions for management, treatment and transportation may also hinder the adoption of practices related to the use of manure, agro-industry by-products or sludge.

Environmental constraints to the adoption of management practices are mainly related to pollution (e.g., heavy metal accumulation through sludge use or by flooding water management, Klee et al., 2004; Uraguchi and Fujiwara, 2012; increased application of herbicides by no-tillage, Annet et al., 2014) and health issues (e.g., for liquid manures, Cole et al., 2000; or by-products without pre-treatment applied to crops). Other environmental constraints can be associated with risk of fire due to leaving crop residues on the soil surface (Luna et al., 2012).

Nonetheless, except for environmental constraints, most of the barriers can be overcome by long term monetary savings or gains associated with the practice. Most practices reduce the need of exogenous N fertilizer, which is one of the main expenses for farmers (Aizpurua et al., 2010; Abalos et al., 2014a; Aguilera et al., 2015a, b). Improved irrigation technology, fertigation, or use of crop residues and agro-industry by-products (Jordan et al., 2010) can reduce crop water requirements, whereas crop rotations and improved irrigation technology may also decrease the need for pesticides and/or herbicides. Conservation tillage practices also reduce labor costs and fuel consumption (Sánchez et al., 2016), while improved irrigation technology and fertigation save time and labor costs (Thomson et al., 2000). In other cases, the practice improves soil quality and can increase crop yields and/or quality in the medium or long term, as for the substitution of synthetic fertilizers by slurry (Plaza-Bonilla et al., 2014a), the use of crop residues and agro-industry by products, fertigation and improved irrigation technology (Ayars et al., 2015; Kennedy et al., 2013), low/no tillage, cover crops and crop rotations (Ferrio et al., 2007). Finally, in some cases, an extra benefit is produced, as for crop residues and agro-industry by-products (Arthurson, 2009).

Further, there are increasing numbers of innovative farmers and associations who are implementing some management practices with positive results and are demonstrating their effectiveness, and advising to other interested or neighboring farmers. Some of the practices are already included in the greening requirements of the European Union Common Agricultural Policy (e.g., crop diversification, crop rotations particularly those which involve legumes), allowing economic incentives to encourage implementation among farmers (Ingram et al., 2015).

5. 2. Assessing policy options to regulate the implementation of different mitigation strategies

The main outcomes of the literature review and the expert judgment as discussed during a workshop held in Butrón (Bizkaia) in December 2016, to synthesize the most promising measures to abate N_2O from cropping systems is presented in Table 1 and 2. This information enabled to perform an assessment based on the simple framework developed by Pannell (2008). This framework was used for choosing environmental policy options to regulate the implementation of different mitigation strategies (Figure 2). In the diagram, the public benefit in the "y" axis refers to the percentage of mitigation (i.e., scale from -100 to 100%) of every mitigation strategy based on the collected literature review values.

We calculated the private net benefit to the farmer on the "x" axis according to the experts' weights (i.e., scale from -5 to 5) on the potential cost and benefit of every mitigation strategy (Table 1). When applying the framework, the use of agricultural extension is highly recommended to engage farmers to adopt strategies that do not imply a cost to the farmer, but that can have large benefits to society (e.g., adjusting N fertilization, manure fertilization, fertigation, increasing legumes, advanced irrigation technology, judicious crop residue management; see Figure 2). The agricultural extension option may include the increase of agricultural demonstrations and communications, to transfer scientific and technological findings to the farming community. This would enhance access to technical education on management practices that deliver mitigation, and support the enlargement of farming networks. Strategies such as injection of slurries, cover crops, application of composted sewage sludge, biochar and use of nitrification or urease inhibitors showed negative or negligible economic net benefits for the farmers (private benefits). In this case, two potential policy options might be applied, according to Pannell (2008): i) positive incentives if societal net benefits are high; and ii) technology development or no action when the public net benefits are moderate or not high enough to warrant incentives (Figure 2).

6. Beyond the plot scale: assessing the combined effect of reduced fertilization and drip irrigation on GHG emissions

Selected management actions show a strong potential for mitigation of specific GHGs. However, it is important to assess this potential within a context of the total GHG budget, including all the involved processes in the production chain, beyond the plot scale, in order to identify possible trade-offs.

Here we present the results of a simple exercise to illustrate these trade-offs by comparing the total life cycle emissions (including infrastructure production, electricity production, fertilizer production, and direct and indirect N_2O emissions) associated with irrigation and N fertilization inputs in a series of hypothetical scenarios. The scenarios cover three irrigation technologies (drip, sprinkler and furrow) under two fertilization application rates (100 and 300 kg N ha^{-1} yr^{-1}) and two levels of water pumping height (surface and 200 m underground) under Mediterranean conditions (Figure 3). We estimated GHG emissions employing published emission coefficients for each process involved, including specific direct N_2O emission factors for Mediterranean irrigation types (Cayuela et al., this issue). Drip irrigation leads to lower overall N_2O emission levels only under certain conditions, particularly when a high energy input has to be applied for water lifting and N is applied at the high rate, as a result of lower water demand and lower N_2O emission factor (Figure 3). However, in some situations, the higher infrastructure burden and the energy needed for pressurizing lead to higher GHG emissions (CO_2 eq ha^{-1} yr^{-1}) from drip irrigation than from furrow irrigation. Likewise, furrow irrigation delivers the lowest emission level when water is easily available and N is applied at the low rate, but the highest when water is extracted from deep wells. On the other hand, our calculations show that the outside-farm production of major inputs such as electricity and N fertilizer is the main contributor to the balance in most situations explored, suggesting that the main focus for reducing the GHG balance of these systems should focus on reducing the CO_2 eq. foot-print associated to these inputs. This could be achieved by reducing the amount of inputs, e.g. optimizing N fertilizer rate and avoiding water with high extraction costs. A complementary strategy would be to minimize the CO_2 eq. emissions from the production of these inputs by, for example, substituting synthetic fertilizers by organic sources of N (residues, biological N fixation) and employing renewable energy for electricity production. It is, therefore, important to consider all the life cycle emissions under each specific circumstance in order to select the best set of practices to maximize mitigation benefits and reach cost-effectiveness in producing a unit of food.

7. Structural changes: behaviors and practices that can function alongside agronomic GHG mitigation

Even if an optimized set of practices in terms of GHGs emissions from soils is implemented, this could still result in an increased overall sectorial emission due to energy intensive practices (such long term transport) or increased waste along the production chain. Globally, 2.7 Tg of N are emitted to the environment in the production of food waste (Grizzetti et al., 2013). A reduction of food waste could significantly reduce the amount of reactive N emitted to the environment during primary production, including N_2O (Bodirsky et al., 2014; Lamb et al., 2016). Between 3 and 15% of N_2O emissions could be suppressed by avoiding food waste at the consumer level (Grizzetti et al., 2013; Vanham et al., 2015). Additionally, curbing food waste would help to avoid GHG emissions associated with waste management, particularly landfill CH_4 emissions, which, in a Mediterranean country such as Spain, represented a similar level of emissions as enteric fermentation by livestock in 2012 (MAGRAMA, 2014). This mitigation measure is not specific to the Mediterranean region, other than considering the relatively high food waste rates that are particularly relevant at the consumption level of the Mediterranean countries belonging to Europe or N America (Gustavsson et al., 2011). While the consumer part is behavioral, the waste produced at other levels, namely supermarket, distribution, agroindustry or farm, can be associated with prices and competitiveness strategies (Parfitt et al., 2010). Food waste reductions could be influenced by policy measures, but diverse conflicts of interest could represent a barrier to implementation.

Changes in diet among population in developed and emerging economies have led in recent years to unexpected increases in GHGs emissions due to increased demand for meat and other livestock products. A reduction of animal protein consumption by 50% in the EU would lead to a reduction of GHG emissions by 25 to 40%, depending on the alternative use of the land (Westhoek et al., 2014). In several Mediterranean countries such as Spain, Italy and Greece, the share of animal protein in the total protein intake has increased from ~35 to over 60%, evolving from a typical Mediterranean diet to a diet rich in animal protein, over recent decades (Lassaletta et al., 2014c). A reduction of 40% of meat and dairy consumption would reduce GHG emissions by 20 to 30%.

Transport of food can also contribute significantly to the footprint of agricultural products. With the exception of France, all countries of the Mediterranean basin are net importers of agricultural products, particularly in the form of feed. In 2009, the countries of the Mediterranean basin net-imported 2.3 Tg N embedded in traded commodities, most of them cultivated in South America, North America, Northern European Countries and Russia (Figure 4). The production of feed in other countries generates at the same time and spatial leakage of emissions that are not considered by the national inventories (Lassaletta et al., 2014a). On the other hand, reducing feed demand within Mediterranean countries could reduce the need for land expansion at global scale. The reintroduction of the Mediterranean diet (i.e., a back reduction to ~35% of animal protein, see Bach-Faig et al. 2011 for a detailed description of the current Mediterranean diet) would reverse this trend: animal production would be lower, land would become available for other purposes and GHG emissions could be reduced by more than 50% (Sáez-Almendros et al., 2013).

In summary, even if the most cost-effective practices are implemented in feed and livestock production, their impacts on GHGs mitigation may be offset by increased demand of high GHG-intensity products (such as meat), increases in food waste at the consumer level and long distance transport. Both reduction of food waste and animal protein consumption represent a reduction of the food demand, and will not only reduce GHG emissions in the agriculture sector, but will also lead to important co-benefits such as decreased demand of agricultural land, giving space for afforestation and reducing deforestation of natural forests, reduce biodiversity loss and improving ecosystem services.

Finally, disconnection between feed and livestock production systems at the regional and global scales results in low nutrient use efficiency of agro-ecosystems, because of difficulties in closing nutrient cycles. Lack of manure in specialized cropping areas leads to higher needs of synthetic fertilizers, and overuse of manure often occur in areas with high animal concentrations (Bai et al., 2014; Billen et al., 2013; Lassaletta et al., 2014a; Naylor et al., 2005; van Grinsven et al., 2014). This phenomenon is driven by the economic benefits associated with spatial concentration of livestock systems, in combination with the low economic value of manure per unit of mass. It has been observed for example in Spain and Italy where areas of livestock production concentration generates too much manure and slurries that are difficult to manage (Lassaletta et al., 2012; Penuelas et al., 2009). In addition, very high manure application rates, typical of livestock concentration areas, are associated with unusually high N_2O EFs under Mediterranean conditions (Heller et al., 2010). The potential of reconnecting livestock and crop farming for mitigating GHG emissions is illustrated by several examples at the local or regional level (Granlund et al., 2015; Sasu-Boakye et al., 2014; Soussana and Lemaire, 2014). Note however that, due to the high level of animal protein production in some regions, both for local consumption and export, a generalized transition to reconnection based on higher local feed consumption would only be possible if it were accompanied by a reduction of animal protein in the human diet (van Grinsven et al., 2015, 2014; Westhoek et al., 2014). A higher demand of non-oil crops feeds replacing soy, with lower protein contents, could otherwise entail a higher land demand that could offset the mitigation benefits or/and could compete with human food. In several Mediterranean countries with livestock production highly dependent on feed imports, a generalized reconnection would require a transition towards the Mediterranean diet and also a reduction of food waste. Thus, important positive synergies between dietary changes, food waste reduction, production close to consumers and livestock-crop reconnection could arise when developed simultaneously.

8. Conclusions

The framework for GHG mitigation provided here, based on solid and comprehensive scientific evidence, is of wide societal, environmental and economic interest, affecting all stakeholders in the Mediterranean agricultural sector, from farmers to governments.

Efficient implementation thus will require effective policies, closer collaboration between scientists, stakeholders and farmers, and enhanced public awareness and engagement.

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Figure captions

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

Figure 2. Policy options based on the Pannell (2008) framework for the GHG mitigation strategies in Mediterranean areas. This is based on choosing environmental policy options to regulate the implementation of different mitigation strategies. The societal public benefit in the y-axis refers to the percentage of mitigation (i.e., scale from -100 to 100%) of every mitigation strategy based on literature review values. We calculated the private net benefit to the farmer in the x-axis according to the weights (i.e., scale from -5 to 5) on the potential cost and benefit of every mitigation strategy. These values were assigned by experts' judgement.

Figure 3. Estimation of greenhouse gas emissions (kg CO_2eq ha^{-1} yr^{-1}) associated to irrigation and N fertilization in Mediterranean cropping systems for three different irrigation types (drip, sprinkle and furrow) under two levels of N fertilization rate (100 and 300 kg N ha^{-1} yr^{-1}) and two levels of pumping height (0 m and 100 m). Emission values are based on data from: infrastructure: Lal (2004); electricity: direct electricity consumption from Aguilera et al. (2015c) and electricity emission factor from Aguilera et al. (2015b); fertilizer production (average N fertilizers, Europe): ecoinvent Centre (2007); N_2O – indirect: IPCC (2006); N_2O – direct: Cayuela et al. (this issue).

Figure 4. Net protein fluxes (expressed in nitrogen) of food and feed imported to Mediterranean regions from the otherworld countries in 2009. Mediterranean regions marked in black. Green countries are those which are net exporting N to the analyzed region. Yellow/red countries are those which are net importing N from the analyzed region. Arrows show fluxes between any region and the studied region. Fluxes below 50 Gg N are not represented. Calculated following Lassaletta et al. (2014b).

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Tables

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Table 1. GHG mitigation performance, costs and benefits and side-effects of agronomic practices in Mediterranean cropping systems

Group of measures	Mitigation measure	Direct GHG abated	% of mitigation	Potential cost (2)	Potential benefit (2)	Potential positive and negative side-effects (3)				
								Other pollutant on farm		
						GHG mitigation out farm	GHG increase outside the farm	Reduced pollutant	Increased pollutant	Crop yield change on farm
Agronomic measures (1)										
Optimal fertilization	Adjust N fertilization to crop needs	N_2O	30-50	**	*****	Indirect N_2O		NO_3^-, NH_3		No effect
	Fertigation	N_2O	30-50	***	****	Indirect N_2O		NO_3^-		Increase
	Substitute synthetic fertilizers by manures	N_2O	20-50	**	****	Indirect N_2O, CO_2	CH_4	P, NO_x, C sequestration	NH_3, heavy metals	No effect
Manures and slurries	Injection of slurries	C seq.	0-10	****	**	Indirect N_2O		NH_3	NO_3^-, CH_4	Decrease
	Immediate incorporation of manures after application	C seq./ N_2O	0-10	**	**	Indirect N_2O		NH_3	NO_3^-, CH_4	Increase
Inhibitors	Use of nitrification inhibitors	N_2O	30-50	****	***	Indirect N_2O	CO_2[c]	NO, NO_3^-	NH_3	Increase[a]
	Use of urease inhibitors	N_2O	30-60	****	***	Indirect N_2O	CO_2[c]	NO, NH_3		Increase
Crop Rotations and cover crops	Cover crops	C seq.	0-10	**	***	CO_2[c]/ Indirect N_2O		NH_3, NO_3^-, P		Variable
	Crop Rotations	C seq.	-	*	***	CO_2[c]		-	-	Increase

Irrigation	Improved Irrigation technology	N_2O /CH_4[b]	50-70	**	***	Indirect N_2O		NO_3^-	NO, CH_4[b]	Increase
Soil tillage	Low/no tillage	C seq.	-	**	***	CO_2[c]		NO_3, NH_3	N_2O	Increase
Crop residues and agro-industry by-products	Crop residues mulching	C seq.	50-70	*	**	CO_2[c]		NH_3		Long-term increase
	Crop residues incorporation	C seq.	50-70	*	*	CO_2[c]		NH_3	CH_4	Long-term increase
	Use of by-products	C seq.	50-70	*	**	CO_2[c]		NH_3		Long-term increase
Composted sewage sludge	Application of composted sewage sludge	C seq.		***	***	CO_2[c]		Heavy metals	CO_2, NO	
Biochar	Use of biochar	C seq./ N_2O	0-50	***	***	Indirect N_2O, CO_2	CO_2[c]	NO_3^-, Heavy metals		Variable
Structural measures (1)										
	Reducing food waste	C seq./ N_2O			***	GHG indirect		NO_3^-, NH_3; NOx		Non-applicable
	Reduction of animal protein consumption	C seq./ N_2O	20-30		***	GHG indirect		NO_3^-, NH_3, NO_X		Non-applicable
	Reconnect crop and livestock areas	C seq. / N_2O	Variable			GHG indirect		NO_3^-, NH_3		Variable

[a] DMPP appears not to affect yield but DCD may provide a slight yield increase (5-10%); [b]CH_4 oxidation favoured; [c]CO_2 due to energy consumption/transport; [d] emissions in paddy soils due to straw addition.

(1) Agronomic (existing or new technical implementation), or structural (change in social behaviour or agronomic organization); (2) Costs and benefits: From 1 (very low) to 5 (very high), based on expert judge and existing literature; (3) Potential positive and negative side effects.

Tables

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Table 2. Constraints to management practice change

Agronomic Measures	Overall (1)	Constraints: Technical	Economic	Social (2)	Environmental (3)	References
Adjust N fertilization to crop needs	Low	Soil analysis needed to adjust dosage. Need to know adjusted crop requirements	Potential increase in labour costs (e.g. split application) and soil analysis	Perception of decreased productivity	N.A.	Aizpurua et al. (2010); Sánchez et al. (2016); SmartSOIL (2015)
Substitute synthetic fertilizers by manures and slurries	Medium	Need to know adjusted crop requirements Need of adequate equipment (for incorporation of slurries)	Transport and application costs New equipment	Legal restrictions (EU Nitrates Directive 91/676/EEC) - (i.e., use, management, treatment and transportation) Bad smells Only applicable to areas with mixed farming systems Perception of decreased productivity	Potential pollution and health issues	Ábalos (2013); Berenguer et al. (2008); Cantero-Martínez et al. (2007); Cole et al. (2000); Díaz-Cruz et al. (2003); Feilberg et al (2011); Küçükdoğan et al. (2015); Maguire et al. (2011); Rodhe (2004); Sánchez et al. (2014)
Fertigation & improved irrigation technology	High	New infrastructure associated with conversion Maintenance difficulties (fertigation)	Initial expensive investment costs	Not for all crops	Potential accumulation of heavy metals in crops (i.e., rice)	Ayars et al. (2015); Kennedy et al. (2013); Santos Pereira et al. (2002); Thomson et al. (2000); Uraguchi and Fujiwara (2012)
Nitrification & Urease inhib.	High	N.A.	Increase of fertilization costs	Not widely spread among neighbouring farmers	N.A.	Abalos et al. (2014a); Linzmeier et al. (2001); Timilsena et al. (2015)
Biochar	Low	Lack of experiments at local conditions	Expensive product (2$ per kilo)	Lack of knowledge on how to produce it on-site; Lack of regulations	N.A.	Hussain et al. (2016)
Composted sewage sludge	High	Access/availability to/of materials	Transport and management	Specific knowledge required to adjust rates to crop requirements and pollution targets Legal restrictions (i.e., Council Directive 86/278/EEC (CEC, 1986); Landfill Directive 99/31/EC (CEC, 1999)) Bad smells and negative image in some areas.	Pollution issues, sanitary problems (antibiotics) and increase in soil salinity	Klee et al. (2004); Threedeach et al. (2012)
Crop residues & agro-industry by-products	Medium	Access/availability to/of materials	Initial investment cost (machinery) Loss of revenue from straw sales	Specific knowledge required (compost) Regulation of rates Strong traditions avoiding the use of by-products (other uses)	Risk of fire (from residues) Sanitary problems (by-products)	Aguilera et al. (2015a); Di Giacomo and Spinelli and Picchi, (2010); Luna et al. (2012); Sánchez et al (2016) ; Taglieri, (2009)
Low/no tillage	Low		Initial investment,	Training and advisory support		Annet et al. (2014), Ingram et al.

		Possible weeds and compaction problems	income loss at short-term, cost on machinery Need of herbicide	Strong traditions of conventional farmers Reluctance from sales technicians	Potential pollution (herbicides)	(2014); Sánchez et al (2014; 2016); Sánchez-Girón et al. (2004); SmartSOIL (2015)
Cover crops	Low	Higher requirements on planning Limited under water scarcity (i.e., water or nutrients competition)	Extra sowing and killing costs associated to the cover crop	Lack of training (e.g, species selection, residue management, kill date) Strong traditions of conventional farmers	N.A.	Alonso-Ayuso et al. (2014); Gabriel et al. (2012); Ingram et al. (2014); Sanz-Cobena et al., (2014b); Sánchez et al (2014; 2016); SmartSOIL(2015)
Crop Rotations	Low	Higher requirements on planning and advice	Loss of market opportunities	Lack of training on selecting crop species and sequences or weed control	N.A.	Ingram et al. (2014); Sánchez et al (2014; 2016); SmartSOIL (2015)

(1) Overall constraints assessment. This has been determined by assessing all specific constraints in an expert judgement analysis; (2) Legal and behavioural; (3) Pollution and sanitary; N.A.: low or non-existing constrain

Tables

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Table 3. Main component and mitigation practices associated of each cropping system in Mediterranean areas (NA = not applicable)

	Main component of radiative forcing		**Main mitigation practice**		**Other pollutants**	
Crop type	***Rain-fed***	***Irrigated***	***Rain-fed***	***Irrigated***	***Rain-fed***	***Irrigated***
Herbaceous	Machinery/external inputs; C seq. (NT)	N_2O	Reducing fuel consumption and external inputs, reduced tillage, crop rotations (including legumes), adjusted N rates, Nis	Water management (e.g. drip irrigation), N fertilization (e.g. adjusted N rates, Nis)	Increased NH_3	Increased NH_3, NO_3^-
Fruit orchards	C sequestration	N_2O	NA	Cover crops, pruning crop residues	NA	NA
Rice	NA	CH_4	NA	Water management, straw management mitigation strategies	NA	Increased N_2O

Figure
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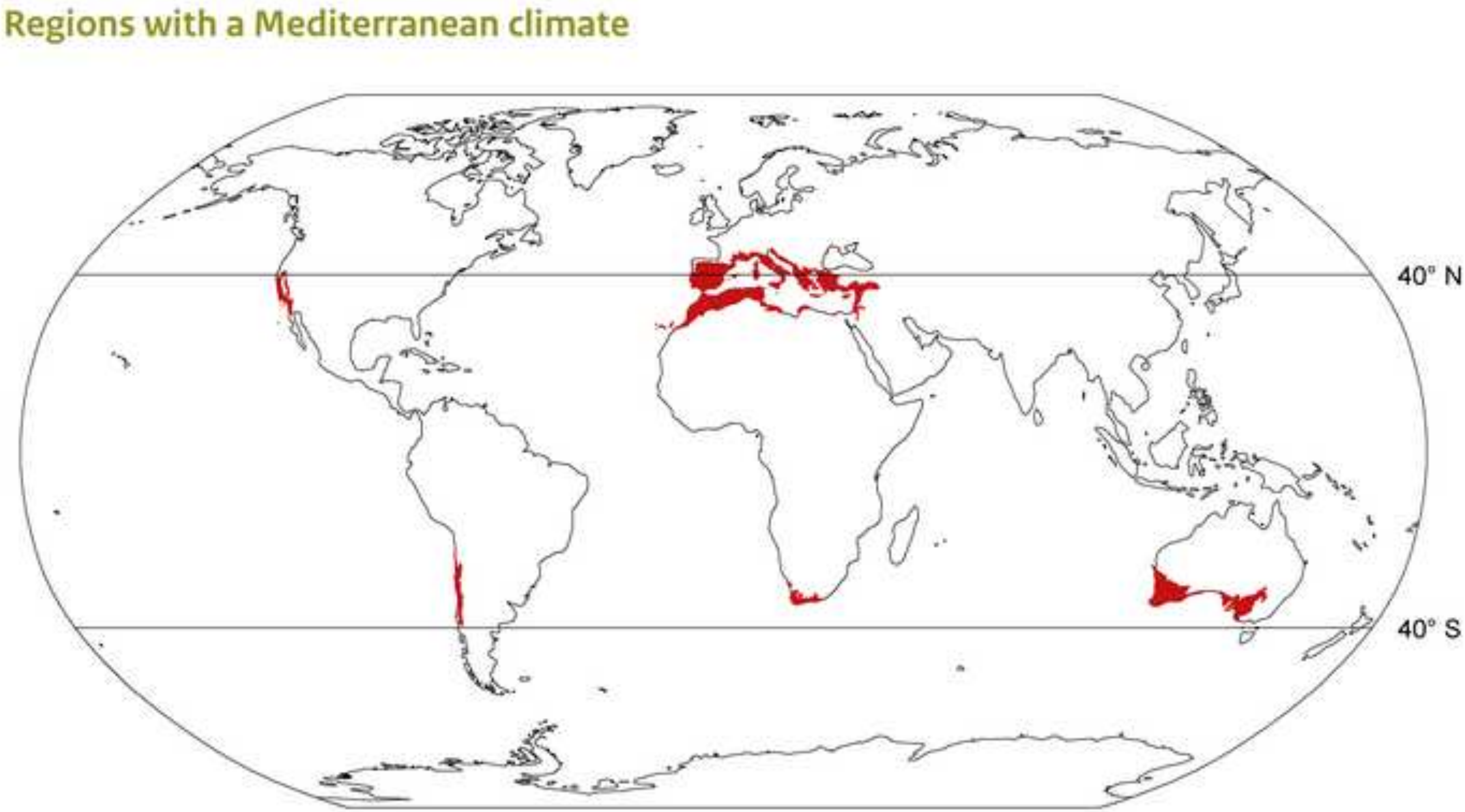

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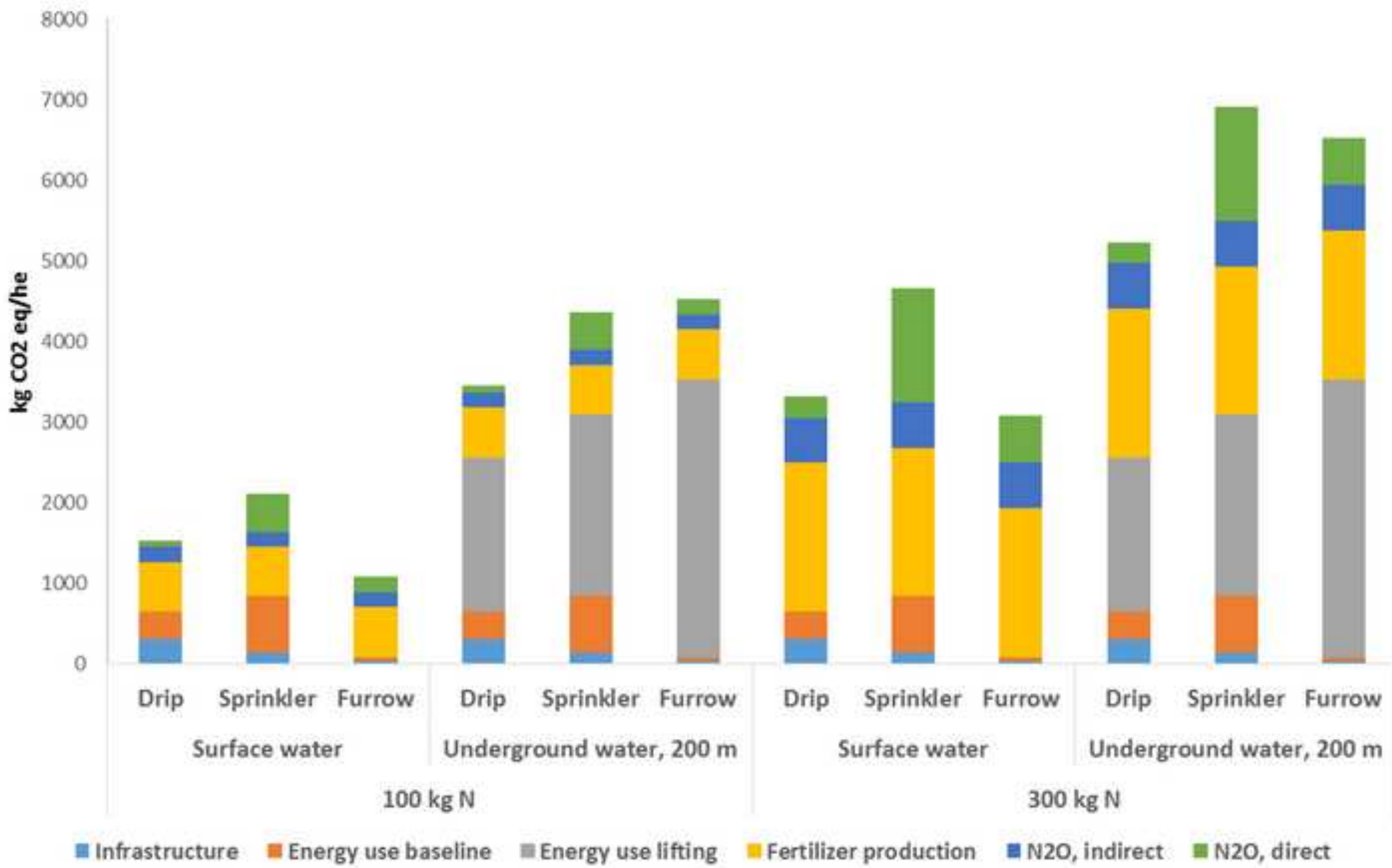

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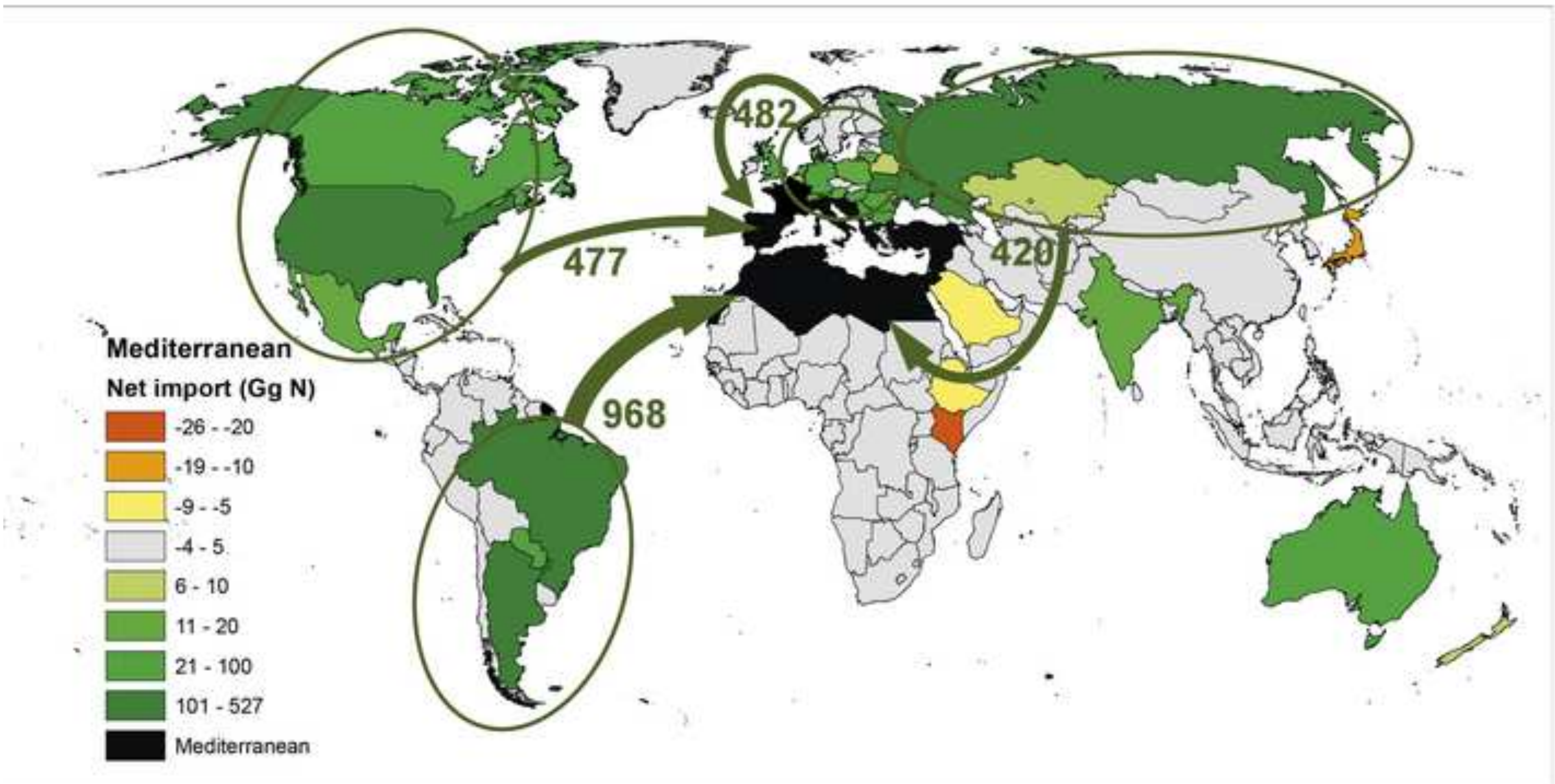

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