Carbon management in dryland agricultural systems. A review

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Abstract

Dryland areas cover about 41% of the Earth´s surface and sustain over 2 billion inhabitants. Soil carbon (C) in dryland areas is of crucial importance to maintain soil quality and productivity and a range of ecosystem services. Soil mismanagement has led to a significant loss of carbon in these areas, which in many of them entailed several land degradation processes such as soil erosion, reduction in crop productivity, lower soil water holding capacity, a decline in soil biodiversity and, ultimately, desertification and hunger and poverty in developing countries. As a consequence, in dryland areas proper management practices and land-use policies need to be implemented to increase the amount of C sequestered in the soil.

When properly managed, dryland soils have a great potential to sequester carbon if financial incentives for implementation are provided. Dryland soils contain the largest pool of inorganic C. However, contrasting results are found in the literature on the magnitude of inorganic C sequestration under different management regimes. The rise of atmospheric CO_2 levels will greatly affect dryland soils, since the positive effect of CO_2 on crop productivity will be offset by a decrease of precipitation, thus increasing the susceptibility to soil erosion and crop failure. In dryland agriculture, any removal of crop residues implies a loss of soil organic carbon (SOC). Therefore, the adoption of no-tillage practices in field crops and growing cover crops in tree crops have a great potential in dryland areas due to the associated benefits of maintaining the soil surface covered by crop residues. Up to 80% reduction in soil erosion has been reported when using no-tillage compared with conventional tillage. However, no-tillage must be maintained over the long-term to enhance soil macroporosity and offset the emission of N_2O associated to the greater amount of water stored in the soil when no-tillage is used. Furthermore, the use of long fallow periods appears to be an inefficient practice for water conservation, since only 10-35% of the rainfall received is available for the next crop when fallow is included in the rotation. Nevertheless, conservation agriculture practices are unlikely to be adopted in some developing countries were the need of crop residues for soil protection competes with other uses. Crop rotations, cover crops, crop residue retention and conservation agriculture have a direct positive impact on biodiversity and other ecosystem services such as weed seed predation, abundance and distribution of a broad range of soil organisms and bird nesting density and success. The objective of sequestering a significant amount of C in dryland soils is attainable and will result in social and environmental benefits.

Keywords: biodiversity, climate change, dryland agroecosystems, ecosystem services, livestock, research perspectives, socioeconomic factors, soil carbon sequestration, soil water.

Contents

1. Introduction

Dryland areas are characterized by a low ratio of mean annual precipitation to potential evapotranspiration (ranging from 0.05 to 0.65) and cover about 41 percent of the surface of the Earth (Lal 2004; Middleton and Thomas 1997). The soils of these areas have an inherent low stock of organic carbon (C) due to climatic limitations. On the contrary, they contain a significant amount of inorganic C, of a persistent nature, mainly present in the form of soil carbonates (Denef et al. 2008). Given the almost nonexistent chance for expanding irrigation in most dryland agroecosystems, other ways of land use optimization need to be identified (Hall and Richards 2013).

Mismanagement such as intensive tillage, excessive grazing or elimination of vegetative cover has resulted in the loss of some 13–24Pg C in grasslands and drylands (Ojima et al. 1995), leading to important degradation processes such as soil erosion, loss of ecosystem services and, ultimately, to desertification (Zika and Erb 2009). Desertification has been directly related to global sustainability threats such as malnourishment and poverty and huge economic losses, particularly in dry climate areas (Zika and Erb 2009). Currently, dryland areas are facing new challenges such as the impact of climate change on hydrological regimes and net primary productivity, as well as an increasing human population pressure (Mouat and Lancaster 2008).

In spite of the limitations and negative perspectives for the future, soils in dryland areas have a great potential to sequester C if appropriate management and land use policies are applied (FAO 2004; Lal 2001; Marks et al. 2009) within an ecological intensification framework (Figs 1 and 2). That framework advocates raising yields without negatively affecting the environment (Cassman 1999). The maximization of soil organic carbon (SOC) stocks in dryland areas not only has the potential to mitigate current increase in atmospheric carbon dioxide (CO_2) concentration, but also can improve soil quality attributes such as aggregate stability, fertility, and nutrient cycling, among others. Those attributes would lead to the reduction of soil susceptibility to degradation processes such as erosion and to the maintenance of agricultural productivity and ecosystem services. This last aspect is paramount to improving the livelihood of people living in drylands, over 38% of the global human population (Maestre et al. 2012).

In the last few decades there has been extensive research in dryland areas regarding soil C sequestration. Various reviews have analyzed soil management and land use practices that maximize C sequestration in dryland systems (Follett 2001; Lal 2002, 2004). However, basic aspects remain poorly understood. In this review we cover key issues related to C management for soil C sequestration in dryland areas, highlighting future research priorities to clarify current knowledge gaps under a multidisciplinary point of view (Fig. 3).

2. The need for carbon management improvement in dryland agroecosystems

2.1. Better understanding of agricultural management and soil carbon issues

2.1.1. Soil erosion and carbon losses

Dryland environments are usually prone to soil erosion due to the lack of a significant soil cover, which is usually aggravated by the high intensity of rainstorms (typical in some dryland areas such as the Mediterranean basin), a reduced soil structural stability, which is generally associated to a limited amount of SOC, and a high human pressure. Other factors such as the presence of steep slopes also exacerbate the susceptibility to soil erosion in drylands (García-Ruiz 2010). Moreover, as a consequence of climate change, some projections suggest that erosion rates could increase by 25-55% during the 21st century (Delgado et al. 2013). In turn, the erosion of soil surface layers can also lead to the exposure of carbonates to climatic elements and acid deposition, aspects that could increase the loss of C from soils to the atmosphere (Lal 2004; Yang et al. 2012).

Three main mechanisms explain the flux of organic C between soil and the atmosphere as a result of an erosive process: (i) at eroding sites SOC is decreased because plant inputs are decreasing with decreased productivity; (ii) SOC decomposition is enhanced due to physical and chemical breakdown during detachment and transport; and (iii) decomposition of the allochthonous and autochthonous C fraction buried is reduced (Van Oost et al. 2007).

In dryland areas, the critical role played by vegetative covers on soil erosion reduction and SOC maintenance has been long recognized. However, in these areas, conventional management techniques hinder the presence of an adequate protection of the soil surface: (i) the use of intensive tillage in herbaceous and tree crops (Álvaro-Fuentes et al. 2008); ii) feed needs for animal production (López et al. 2003); (iii) excessive grazing (Hoffmann et al. 2008); and (iv) the recent high feedstock demand for bioenergy (Miner et al. 2013). In developing countries of Asia and Africa, the extractive nature of using crop residues as fodder for cattle and animal dung as a cooking fuel poses a serious problem to soil quality and the sustainability of crop production (Lal 2006). In those countries, soil organic carbon decline needs to be counteracted by increasing the amount of crop residues produced. However, due to the highly weathered nature of soils in some developing regions such as West Africa some fertilization is needed to avoid the depletion of soil nutrients (Bationo et al. 2000).

Obviously, there is a need for a reliable economic assessment of the long-term benefits of maintaining crop residues on the soil surface in terms of C sequestration, erosion reduction, nutrient cycling and water retention. This information would be of a great value for farmers in order to reduce the amount of crop residues that is currently removed from agricultural fields given the concomitant short-term economic returns of this practice.

The use of conservation tillage and more recently no-tillage practices leave the soil covered by crop residues, which has long been recognized as an excellent means of decreasing soil erosion (Delgado et al. 2013). For instance, given their potential in reducing soil degradation, the Chinese government is promoting the use of conservation tillage practices throughout vast dryland regions of northern China (Wang et al. 2007). According to data from the Chinese national projects regarding conservation tillage, the last authors reported a 60 to 79% decrease in soil erosion when using no-tillage. Similarly, in a modelling study, Fu et al. (2006) reported a decrease of soil erosion from 17.7 to 3.9 t ha^{-1} yr^{-1} when adopting no-tillage, due to mitigation of rill generation. Different tillage experiments have been carried out by the International Center for Agricultural Research in the Dry Areas (ICARDA) in the Central Asia region. According to Thomas (2008), those experiments show that conservation tillage performed well in terms of energy and soil conservation and that crop yields were either not affected or slightly increased. Unfortunately, the benefits of no-tillage have not been tested in all the dryland agricultural areas of the world. For instance, in Central Asia, only Kazakhstan has a brief history in adopting no-tillage farming with locally manufactured machinery (Thomas, 2008). The study about the potential use of no-tillage in Africa carried out by the German Agency for Technical Cooperation (1998) concluded that in the semiarid and arid regions of West and Southeastern Africa different constraining factors such as (i) short growing season, (ii) low levels of biomass production and (iii) competition for crop residues would make more viable the use of reduced tillage methods. Similarly, for semiarid West Africa, Lahmar et al. (2012) concluded that is unlikely that conservation agriculture practices, which are based on the presence of crop residues on the soil surface, will be adopted by farmers due to the competition with other residue uses.

Recent technological advances can improve the performance of no-tillage in dryland areas. For instance, in field crop production, the development and use of stripper-headers as attachments for combines has a great potential to reduce soil erosion risks when no-tillage is used. This technological improvement virtually leaves all crop residues on the soil surface, thus reducing harvest costs by lower fuel consumption (Spokas and Steponavicius 2011) and, as a result, diminishing CO_2 emissions to the atmosphere. This technology is also of great interest in areas that receive winter snow for its capacity to trap the snow (Henry et al. 2008). Moreover, the presence of taller vertical crop stalks reduces the wind speed, thus lowering the chance of losing soil C due to wind erosion and minimizing water evaporation (Henry et al. 2008).

Soil management in tree-cropping (e.g. vine, olives, almonds, etc.) traditionally involves frequent tillage because uncontrolled weed growth competes for water resources with crops. However, some studies have shown that soil erosion can be minimized while maintaining yields with the use of a properly managed vegetative cover (Gómez et al. 1999; Kairis et al. 2013). In this context, more research is needed to find the optimum technological choices for cover cropping in order to enhance SOC stocks while reducing the susceptibility to soil erosion under water-limiting environments. This would imply the identification of (i) the best species to act as vegetative cover, (ii) optimum termination strategies such as chemical weeding or physical clearing, and (iii) the best dates for termination according to local rainfall distribution and crop water needs.

Future research also must address the impacts of the demand for cellulosic-based fuels on soil conservation and SOC stocks maintenance (Wilhelm et al. 2007). In this line, Miner et al. (2013) modeled the impact of harvesting crop residues for biofuel production, in a wheat-corn-fallow cropping system in the semiarid central Great Plains. These authors observed unsustainable wind erosion rates after harvesting 10 to 30% of corn residues, while up to 80% of wheat residues could be removed without reaching the tolerable soil loss limit. However, they also found that any removal of wheat or corn residues implied a loss of SOC. This study clearly indicates that the use of crop residues for bioenergy needs to be considered with caution in dryland areas. Similarly, in grassland systems, the management of livestock grazing intensities needs to be optimized to reduce soil compaction and surface sealing, processes that can exacerbate the loss of SOC by wind and water erosion and reduce the production of biomass (Delgado et al. 2013). For instance, in these systems it has been reported that erosion can lower soil productivity by at least 10-20% due to a reduction of SOC and nutrients and to related negative impacts on other soil properties (Delgado et al. 2013). In developing countries, the lack of affordable nutrients and soil mining makes crops entirely reliant on soil organic matter (Samaké et al. 2005).

Current research on the effects of agricultural management practices on soil erosion and C stabilization has been performed at the plot scale. For that reason, the role of erosion-deposition processes on SOC balance at the landscape scale has not been accurately assessed (Govaerts et al. 2009; Izaurralde et al. 2007). This would also help us clarify the current controversial and site-specific effects of soil erosion on the global C cycle (Kuhn et al. 2009) without forgetting the pool of inorganic C. Currently there is a lack of understanding regarding the impact of wind and water erosion on greenhouse gas emissions (Kuhn et al. 2012), mainly methane (CH_4) and nitrous oxide (N_2O). For instance, erosion can increase indirectly N_2O emissions in upper slope landscape positions due to the greater application of nitrogen (N) fertilizers carried out by the farmers to compensate for the reduction in soil fertility. In dryland ecosystems the maintenance of a protective vegetative cover appears as the most practical and straightforward strategy to reduce soil C losses by erosion. Consequently, agricultural activity in those areas must be based on conservation agriculture practices, leaving crop residues on the soil surface.

2.1.2. Soil inorganic carbon sequestration and dynamics

There is a growing recognition that the interaction of agricultural practices and soil inorganic carbon is of key importance to the global C cycle. However, the lack of information on soil inorganic carbon dynamics in cropland soils as affected by land use and management, as well as the uncertainties regarding pedogenic inorganic C in relation to soil inorganic carbon sequestration, were identified in the late 90's as major knowledge gaps regarding the C sequestration potential of agricultural activities (Lal and Kimble 2000). These authors pointed out the need to quantify the dynamics of the soil inorganic carbon pool in dryland soils of arid and semiarid regions and proposed several land use and soil management strategies for soil inorganic carbon sequestration in dryland ecosystems, through the formation of secondary carbonates. Through the latter process, Lal (2004) reported an average soil inorganic carbon sequestration rate of 0.1-0.2 Mg ha^{-1} yr^{-1} in dryland ecosystems.

Apart from its potential as atmospheric CO_2 sink, soil inorganic carbon may play an indirect positive role in soil aggregation through the interaction between carbonates and soil organic matter. According to Bronick and Lal (2005), the beneficial effect of carbonates on soil structure is regulated by soil organic matter. At low organic matter contents, the water stability of soil macroaggregates is strongly correlated with the carbonate content (Boix-Fayos et al., 2001). Carbonates can also contribute to soil organic matter protection and stabilization. In calcareous soils, with high exchangeable Ca, high carbonate contents enhance physical SOC protection within aggregates due to a cation bridging effect that leads to slower SOC decomposition rates compared with non-calcareous soils (Clough and Skjemstad 2000). However, depending on soil management, the relative role of carbonates and soil organic matter in soil aggregation may alter the aggregates hierarchy as observed by Virto et al. (2011) in carbonate-rich soils in semiarid Spain.

However, in the last decade few studies have evaluated the impacts of land use and management practices on soil inorganic carbon dynamics in semiarid lands (Denef et al. 2011). In some of those studies soil inorganic carbon storage has proven to be significantly higher in cultivated dryland soils compared with native grassland soils (Cihacek and Ulmer 2002; Denef et al. 2008), but the reduction of tillage may have differing effects in the long-term. Hence, contrasting results have been obtained when comparing the amount of soil inorganic carbon under different types of tillage (Blanco-Canqui et al. 2011; Moreno et al. 2006; Sainju et al. 2007).

Carbon sequestration as inorganic forms has been proposed as a viable alternative in irrigated soils in semiarid and arid regions (Entry et al. 2004). However, the literature on this issue is scarce and also with contrasting arguments and results. Hence, while some authors consider that secondary carbonate precipitation is an important mechanism of soil C sequestration, other argue that dissolution of carbonates should be considered sequestration (Sanderman 2012). In this context, when calcium-enriched groundwaters are used for irrigation $CaCO_3$ is formed, thus leading to the release of CO_2 (Schlesinger 2000).

Likewise, the studies on soil inorganic carbon dynamics under long-term irrigated farming have shown mixed results. While Entry et al. (2004) and Wu et al. (2009) reported a greater amount of soil inorganic carbon in irrigated treatments compared with native soils, Denef et al. (2008) did not find significant difference in soil inorganic carbon between irrigated and dryland treatments. In turn, Halvorson and Schlegel (2012) found that under limited irrigation soil inorganic carbon tends to increase with time in all soil depths supporting the results by Blanco-Canqui et al. (2010). In any case, an account of the entire C footprint would be needed when considering soil inorganic carbon sequestration with irrigation, taking into account the energetic cost of pumping water and the concomitant release of CO_2 in the case of pump-based irrigation systems (Schlesinger 2000).

Other studies have linked soil inorganic carbon sequestration with the quality of the irrigation water. For instance, Eshel et al. (2007) found that long-term irrigation of semiarid soils undergo significant losses of soil inorganic carbon in the root zone compared with non-irrigated soils and that these soil inorganic carbon losses are much larger in soils irrigated with potable fresh water compared with effluent-irrigated soils. They concluded that effluent water inhibited carbonate dissolution. Data provided by Artiola and Walworth (2009) suggest that the release and leaching of both SOC and soil inorganic carbon are directly linked to the dissolution of soil carbonates, and therefore related to irrigation water quality. However, the literature on the effects of agricultural land management on leaching of dissolved inorganic C is also limited (Walmsley et al. 2011).

Furthermore, most of the studies dealing with CO_2 emission from agricultural soils to the atmosphere assume that all the CO_2 emissions are due to respiration. Some authors, however, have questioned whether this assumption is valid in calcareous soils. For instance, Tamir et al. (2011) reported that the dissolution of soil carbonates can contribute up to 30% of the CO_2 emitted from calcareous soils in Israel. In contrast, in an incubation experiment, Ramnarine et al. (2012) estimated that the proportion of CO_2 originating from carbonates was up to 74% in both conventional tillage and no-tillage samples from a calcareous soil in Canada. The last findings suggest that the CO_2 emitted by respiration could be largely overestimated in calcareous soils.

The complex nature of the accumulation and depletion processes involved in soil inorganic carbon sequestration might partially explain, not only the knowledge gaps mentioned above, but also the contrasting results found in the literature on the magnitude of soil inorganic carbon sequestration under different management regimes (Rodeghiero et al. 2011). As pointed out by Sanderman (2012), in his recent review on the major soil inorganic carbon transformations in soils as affected by the agricultural management in Australia, more research is needed to determine the real importance that management induced changes in soil inorganic carbon stocks have on net greenhouse gas emissions.

Despite its potential in semiarid and arid regions, the implementation of key practices for soil inorganic carbon sequestration through pedogenic carbonate formation is still impeded by our limited knowledge on this particular issue.

2.1.3. Soil biodiversity and ecosystem services

Biodiversity is considered fundamental for the stability of ecosystem services in agricultural systems (Naeem et al. 2012). Plant biodiversity represented by polycultures, crop rotations, cover crops, and agroforestry with perennial vegetation can provide important ecosystem services (Perfecto and Vandermeer 2008). In agricultural systems, the use of that diversity in combination with other agricultural practices such as vegetative mulches, fertilization, irrigation and the reduction of tillage intensity affects soil C pools, increasing net productivity (Hoyle et al. 2013; Stockmann et al. 2013).

In dryland agroecosystems, the lack of water is the main limiting factor affecting crop diversity, net primary productivity, SOC dynamics and soil microbial activity (Skopp et al. 1990). In dryland agriculture there are four important aspects to improve productivity, provide ecosystem services and increase SOC: (i) taking advantage of plant diversity (i.e. use of legumes, agroforestry); (ii) establishing proper crop residue management; (iii) improving our knowledge about the importance of soil biology on C cycling; and (iv) determining the optimum level of ecological crop intensification (i.e. rotations, fertilization, etc.).

Plant diversity promoted by crop rotations (West and Post 2002) usually increases aerial biomass and favours the diversification of root systems (i.e. belowground C allocation), with a diverse effect on SOC by root derived products (Stockmann et al. 2013). Deep rooting can contribute to the enhancement of soil C stock in depth (Hoyle et al. 2013; Jobbagy and Jackson 2000). In rainfed agriculture the development of practices for efficient use of the whole soil profile, such as the use of species and cultivars with deeper and improved root systems, must be considered, as it is highlighted in section 2.2. The development of better-adapted root systems needs to be accompanied by an improvement in the current knowledge about the changes that occur in soil biodiversity with soil depth and their effects on C cycling (Witt et al. 2011).

Given the low reliability of seasonal precipitation forecasts in semiarid areas, the selection of crops with assured positive net returns is a difficult task (Saseendran et al 2013). The inclusion of legumes in crop rotations has been proposed as a practice for increasing SOC in dryland conditions (Sanderson et al. 2013). Legumes play a positive role in the reduction of subsequent crop fertilization needs. However, the higher mineralization rate of leguminous crop residues can increase the risk of N leaching during fallow periods, since most semiarid dryland systems give small opportunities to the use of cover crops. Furthermore, the addition of N rich crop residues from legumes is not always followed by higher SOC stocks as a consequence of the greater rate of decomposition (Stockmann et al. 2013). Moreover, under a purely economic perspective, the inclusion of legumes in semiarid dryland crop rotations is not always beneficial (Álvaro-Fuentes et al. 2009a) and could also lead to greater N losses as N_2O (Sanderson et al. 2013).

Crop residue properties (i.e. quantity, quality, placement and supply interval) affect SOC and soil fauna, bacteria and fungi (Agren and Bosatta 1996; Dalal and Chan 2001). The amount and composition of crop residues are directly affected by crop species, and also by agricultural practices such as fertilization or irrigation. An increase of crop residues could improve N use efficiency and reduce N losses (Blanco-Canqui 2010). However, as it has been already mentioned in section 2.1.1., under rainfed conditions, the low availability of crop residues reduces the potential for C storage (Blanco-Canqui et al. 2011; Stockmann et al. 2013). As a consequence, in drylands it is important to develop an integrated strategy to maintain and manage crop residues according to plant and soil biodiversity and economics.

The soil microbial community is an indicator of soil quality and soil fertility and its functional diversity and changes deserve further study (Dalal and Chan 2001). The microbial community has the capacity of suppressing the impacts of pathogens (Verhulst et al. 2010) and directly affects SOC dynamics. Moreover, other important indicators of soil biological activity such as earthworm abundance and community composition result in larger and interconnected pores increasing water infiltration (Verhulst et al. 2010), a fact that has a direct effect on C inputs to the soil, microbial activity and SOC decomposition. Other organisms such as arbuscular mycorrhizal fungi play an important role in nutrient acquisition, drought resistance and maintenance of soil stable aggregates (Oehl et al. 2005; Sanderson et al. 2013).

A reduction in cropping intensification decreases species diversity and plant biomass and could lead to the reduction of the loss of natural resources (Tongway and Hindley 2004). In dryland agricultural systems, crop rotations, cover crops, crop residue retention and conservation agriculture increase water use efficiency, biomass production and SOC and have a direct impact on biodiversity and different ecosystem services such as weed seed predation (Baraibar et al. 2011), abundance and distribution of a broad range of soil organisms (Buckerfield et al. 1997; Henneron et al. 2015; Sapkota et al. 2012) or bird nesting density and success (VanBeek et al. 2014). On the other hand, there are some complex interactions that determine crop productivity and C storage in soils, making difficult the observation of real patterns and the development of management recommendations (Corsi et al. 2012). Then, before establishing the degree of ecological intensification to be applied in dryland agroecosystems, it is needed to determine how the interactions between soil microbial diversity, plant communities and cropping practices can improve productivity and affect SOC (Duffy 2009; Zavaleta et al. 2010). The use of various management practices (e.g. polycultures, crop rotations, agroforestry, reduction of tillage, etc.) enhances the positive feedback existing between soil carbon sequestration and biodiversity in rainfed farming systems.

2.2. Adoption of more efficient water management practices

The productivity of dryland agricultural systems is hindered by the water deficit created by the difference between precipitation and potential evapotranspiration. Given the irregularity of rainfall in most dryland areas, there is a strong need to develop regional decision tools to establish the most appropriate agricultural management strategies (i.e. choice of crop, sowing time, management of soil cover, timings and rates of N application, etc.) according to the amount of water held in the soil. Implementing proper decisions would increase the amount of biomass produced and SOC sequestered. To achieve this objective, the information obtained in long-term field trials is essential for improving current knowledge. To increase the amount of biomass produced and, consequently, the above- and below-ground inputs of C to the soil, the amount of plant available water needs to be enhanced. To accomplish this, three factors need to be maximized: (i) precipitation capture; (ii) water retention in the soil; and (iii) crop water use efficiency (Peterson and Westfall 2004). The amount of precipitation captured is strongly related to soil structural stability and to the abundance and continuity of macropores in the soil surface. Agricultural management practices play a major role on the buildup and breakdown of soil surface aggregates (Plaza-Bonilla et al. 2013b), thus directly affecting soil structure. In dryland areas, soil aggregate stability needs to be maximized to guarantee (i) a continuous network of soil macropores and (ii) a durable physical protection of SOC against microbial decomposition. The accumulation of C in the soil surface (i.e. C stratification) as a consequence of the use of different agricultural practices (e.g. no-tillage, biochar addition) usually improves water infiltration and saturated hydraulic conductivity (Franzluebbers 2002; Jordán et al. 2010). Recent advances in X-ray computed tomography are increasing our knowledge about soil structure and the impacts of agricultural management on soil macroporosity (Perret et al. 1999). Other tools such as the measurement of soil sorptivity are used to assess the potential of soil to capture rainfall (Shaver et al. 2013). Nevertheless, with the current knowledge, it is still difficult to develop tools (i.e. models) that quantify with precision the impact of agricultural management on the dynamics of the soil porous network (Pachepsky and Rawls 2003). The development of these models would be of great interest to identify the best practices to capture rainfall in dryland areas as a function of soil characteristics. Another important strategy to enhance the amount of water retained in the soil is rainwater harvesting, which consists in collecting and storing runoff water in shallow troughs. This system is widely used in developing countries and in specific tree-cropping systems in some developed ones (FAO, 2004). A thorough review about the implementation of rainwater harvesting techniques in the sub-Saharan Africa can be found in Vohland and Barry (2009).

Once water has infiltrated into the soil profile, the efforts must be placed on its retention. In dryland areas, maintaining the soil surface covered is critical to preserve water (Montenegro et al. 2013). Different cropping technologies have been proposed in order to increase soil water retention. Traditionally, fallow has been used in dryland areas to increase soil water content, N availability and weed control. Many studies have pointed out the inefficiency of this practice in terms of water storage. Thus, the works by Lampurlanés et al. (2002) and Hansen et al. (2012) showed that only 10-35% of the rainfall received was available for the next crop when fallow was included in the rotation. Water is lost during fallow periods due to evaporation given (i) the low amount of residues covering the soil surface and (ii) the frequent use of tillage to eliminate weeds in most of the dryland agroecosystems. Thus, research has also been oriented to reduce bare fallow periods by intensifying cropping systems and the use of green manures such as legumes. According to Álvaro-Fuentes et al. (2008), the suppression of long-fallowing leads to an improvement of soil structural stability, thus increasing water infiltration and retention. Moreover, when fallow is eliminated, C inputs are increased due to a higher production of biomass which enhances the amount of SOC sequestered (Álvaro-Fuentes et al. 2009b; Virto et al. 2012). However, in areas with a high water deficit, the benefits of using cover crops as green manure are offset by water lost for subsequent cash crops (Hansen et al. 2012). The use of legumes as green manure could also have a detrimental impact on SOC as it has been discussed in the previous section.

The use of conservation tillage systems such as reduced tillage or no-tillage has been pointed out as one of the most promising strategies to enhance SOC stocks in dryland areas due to its beneficial effect on soil water storage (Fig. 1), which results in turn in greater biomass production and higher C protection within soil aggregates (Aguilera et al. 2013a). Significant rates of C sequestration have been reported in different dryland cropping systems when using no-tillage. For instance, Vågen et al. (2005) reported a rate of 0.05 to 0.36 Mg C ha^{-1} yr^{-1} in Sub-Saharan Africa while Farina et al. (2011) reported a rate of 0.31 Mg C ha^{-1} yr^{-1} in a no-till sunflower-wheat rotation in Italy.

However, the general hypothesis that no-till is always followed by SOC sequestration is still controversial since in most of the studies comparing the effects of different tillage systems on soil C, only the surface soil (0-30 cm depth) has been taken into account (Govaerts et al. 2009; Palm et al. 2013). Furthermore, attention has to be paid to a possible increase in the emission of N_2O when using low-intensity soil management systems, as a result of the greater amount of water stored in the soil. That increase could offset the amount of C sequestered under reduced tillage and no-tillage, since N_2O has a global warming potential 298 times greater than CO_2 (Six et al. 2004). However, recent works have found lower N_2O emissions when no-tillage is practiced in the long-term due to a reduction of anaerobic microsites in the soil (Plaza-Bonilla et al. 2014; van Kessel et al. 2013). These last aspects indicate that future research must take into account the whole C footprint associated to the long-term effects of agricultural practices on greenhouse gas emissions in dryland soils, taking advantage of long-term field experiments and properly validated models.

Once retained in the soil, water needs to be used efficiently by plants, a process that can be improved by using a proper crop management and election of plant material. Drought-prone environments need specific breeding programs in order to find traits related to an efficient water use through stomatal transpiration (Blum 2005). For instance, an improved stomatal control, higher photosynthetic rates and increased stay green have been enumerated in new drought-tolerant corn cultivars (Roth et al. 2013). Similarly, the improvement of root systems to enhance water use in dryland environments remains a critical issue (Hall and Richards 2013). The selection for more adapted root systems would also impact positively on C sequestration, since belowground biomass constitutes an essential input of C to the soil, given its longer time of residence compared with the aerial biomass inputs (Rasse et al. 2005). There also is an urgent need to identify genotypes with traits better adapted to no-tillage conditions, such as a more vigorous emergence or a higher resistance to different diseases (Herrera et al. 2013).

Crop water use is significantly affected by other management practices such as crop fertilization, which affects leaf area and transpiration. In drylands, the use of fertilizers is not always followed by an increase of SOC stocks due to the low crop response to the application of nutrients such as N as a consequence of lack of water. As a result, in dryland agriculture the effects of N fertilization on SOC usually appear in the long-term (Álvaro-Fuentes et al. 2012) and still are a controversial issue (Khan et al. 2007), especially if the energy cost associated with the N fertilizer production is taken into account. In this context, the use of organic fertilizers (i.e. slurries or manures), which is a common practice in some drylands, has the potential to increase SOC stocks and C physical protection within soil aggregates (Plaza-Bonilla et al. 2013a). However, this strategy is only applicable in certain developed areas with nutrient surpluses. Another recent work shows a decrease in N_2O emissions when using organic fertilizers in comparison with the use of synthetic products in dryland areas (Aguilera et al. 2013b).

Maximizing soil water availability for plants is of paramount importance in dryland areas for enhancing C sequestration in soils. To achieve this, long bare fallow periods need to be suppressed and soil tillage must be reduced or eliminated.

2.3. Livestock integration into dryland farming systems

The impact of livestock activities on the environment is either direct like grazing (in extensive livestock systems) or indirect through production of forage crops for confined livestock feeding. Presently, livestock production accounts for 70 percent of all world agricultural land and 30 percent of the Earth's land area (Steinfeld et al. 2006). In relation to ecological conditions and environmental changes, the increase in the demand of animal products will affect more intensely grasslands in arid, semi-arid and tropical regions (Follett and Schuman 2005) (Fig. 2). Despite the inherently low SOC sequestration rates that have been reported in grasslands when compared with other land uses, their global impact can be significant given the surface covered by this land use (Follett and Schuman 2005). The potential C storage in grasslands varies according to climatic conditions and management

(Silver et al. 2010). For instance, the last authors reported soil C contents of 200 Mg C ha^{-1} in the first 100 cm soil depth in annual grass-dominated rangelands in California.

Soil C can be affected by more than one process when grasslands are used for grazing: soil compaction, a decrease of standing biomass, diminution of vegetation coverage, changes in root biomass, and potential increases in erosive processes (Jing et al. 2014). Conflicting results have been reported regarding the effect of grazing intensity on SOC. While some authors found an increase in SOC stock with intensively managed grasslands (Conant et al. 2003; Reeder et al. 2004) others concluded that high stocking rates reduce the aboveground grass biomass and, as a consequence, diminish soil C stocks, which affect the labile fractions such as the particulate organic matter (Silveira et al. 2013; Smith et al. 2014). Regarding to this subject, Han et al. (2008) observed a decrease of 33 and 24 % in SOC and total N (0-30 cm depth), respectively, under heavy grazing when compared to light grazing in a semiarid continental steppe in north-eastern Inner Mongolia. These results were confirmed by Steffens et al. (2008), who found a deterioration of different soil properties including organic carbon in a heavily grazed steppe in the same semiarid region. Furthermore, the intensity of grazing can also influence soil inorganic carbon dynamics. Reeder et al. (2004) reported an increase of soil inorganic carbon of 10.3 Mg ha^{-1} in the 45 to 90-cm depth of a heavily grazed treatment compared to its exclosure in an experiment carried out in the Central Plains of the USA. However, in this study the authors were not able to distinguish whether the increase in soil inorganic carbon represented newly fixed C or a redistribution of the existing material.

The type of grazing can also influence SOC content. For instance, the multi-paddock system usually leads to greater C contents than the light continuous system (Teague et al. 2011). A synthesis of the effects of grazing on SOC stocks can be found in the work of Pineiro et al. (2010). Proper grazing management should maintain a favorable C balance in the ecosystem versus haymaking or combined practices (Oates and Jackson 2014; Ziter and MacDougall 2013). For example, the use of conservative practices to avoid overgrazing or to fence plots has represented a solution to erosion damages in Chinese grasslands (Fang et al. 2010; Han et al. 2008).

Domestic herbivores tend to uncouple C and N cycles by releasing digestible C as CO_2 and CH_4, and by returning digestible N at high concentrations in urine patches. The latter aspect is directly linked to the stocking rate and the period of grazing, and can potentially increase the emissions of N_2O (Soussana and Lemaire 2014). The use of short grazing periods or nitrification inhibitors has been reported to lower N_2O emissions from urine patches (Li et al. 2013). However, the effectiveness of nitrification inhibitors is arguable given the spatial and temporal heterogeneity of the urine patches in grazed systems.

The rapid population growth after the II World War and the increase in the demand of animal products has facilitated the transformation of natural vegetation to arable land to produce feed for animals. Traditionally, extensive livestock production was based in local available feed resources such as crop residues and rough vegetation that had no value as human food. The conversion of pastures to arable crops caused changes in soil C distribution due to soil aggregation disturbance and changes in crop residue inputs and decomposability, thus resulting in C losses (Matos et al. 2011; Su 2007). A study conducted in 27 European soils quantified C losses when grasslands were converted to croplands (i.e. a loss of 19±7 Mg C ha^{-1}), and an accumulation of 18±7 Mg C ha^{-1} when cropland was converted to grassland (Poeplau and Don 2013). Similarly, in a study about the potential for soil C sequestration in Central Asia, Sommer and de Pauw (2011) pointed out that the conversion of native land into agricultural land and the degradation of rangelands led to a loss of 4.1% of the total SOC pool. In turn, global warming and drought in grasslands will change the physiology of grassland species and, consequently, the SOC balance (Sanaullah et al. 2014). In Europe (the EU25 plus Norway and Switzerland) some predictions suggest that cropland SOC stocks from 1990 to 2080 would decrease by 39 to 54%, and grassland SOC stock could increase up to 25% under the baseline scenario, but could decrease by 20–44% under other scenarios (Smith et al. 2005).

Current knowledge about the synergies and trade-offs in adaptation and mitigation strategies in grasslands is still limited and requires further research (Soussana et al. 2013). In this regard, three specific actions are suggested: (i) in all cases grazing management should be adapted to increase the resilience of plant communities to climatic variability (Su 2007); (ii) special attention should be paid to the improvement of agro-silvo-pastoral systems (Gomez-Rey et al. 2012); and (iii) natural margins should be considered due to their role in SOC sequestration (D'Acunto et al. 2014; Francaviglia et al. 2014).

2.4. Climate change adaptation and mitigation

In the agricultural and forestry sectors, climate change adaptation refers to the adoption of practices that minimize the adverse effects of climate change, while mitigation deals with the reduction of greenhouse gas emissions from agricultural and animal husbandry sources and the increase in soil C sequestration. Since the mid-18th century, anthropogenic activities have contributed 169 Gt CO_2, 43% of which have accumulated in the atmosphere (IPCC 2013). Raising atmospheric CO_2 levels favours plant photosynthesis and also the reduction in stomatal conductance, which in turn promotes higher water use efficiency (Ko et al. 2012). The increase in water use efficiency may be hindered by the rise in canopy temperature expected under CO_2 enrichment, resulting in higher leaf transpiration (Kimball et al. 2002). Despite this latter process, results from different free-air concentration enrichment (FACE) experiments have demonstrated the positive general effect of rising atmospheric CO_2 levels on plant production, especially in C3 crops (Ainsworth and Long 2005; Long et al. 2006). Likewise, it has been demonstrated that the increase in plant production under CO_2 enrichment conditions has a direct impact on C dynamics, and particularly on long-term SOC storage if accompanied with increased inputs or reduced losses of N, although not all FACE experiments have reported a final increase in SOC (Prior et al. 2005; van Groenigen et al. 2006).

However, under climate change conditions, the C cycle in agricultural systems will not only be affected by the increase in atmospheric CO_2 concentration, but also by the predicted changes in other variables (i.e. amount and intensity of rainfall) and also by the management practices implemented. In particular, for dryland areas, general circulation models predict significant increases in mean surface temperatures and expected decreases in total annual precipitation with both changes in the seasonal distribution pattern and higher occurrence of extreme events (Gao et al. 2006; IPCC 2013). Consequently, in dryland agroecosystems, the predicted changes in climate will likely condition the positive response found in some FACE experiments between CO_2 enrichment and SOC levels (Dijkstra and Morgan 2012; Liebig et al. 2012).

Crop growth and productivity respond to changes in surface temperature. Although this response can be either positive or negative (Wilcox and Makowski 2014), in southern latitudes and semiarid areas acceleration of maturation and/or heat stress due to warming can have negative impacts on crop production (Lavalle et al. 2009), thus offsetting the potential gain in SOC stocks expected under CO_2 enrichment. In some African countries, for example, crop yields could be reduced by 50% by 2020 (Marks et al. 2009). Limited information exists in the literature about the interactive effects of warming and CO_2 increases in C dynamics in agricultural systems. The few available studies show that warming increases SOC losses due to the acceleration of soil organic matter decomposition (Dijkstra and Morgan 2012; Liebig et al. 2012). However, the increase in surface temperatures may also increase soil drying. This is critical in dryland agroecosystems in which soil water availability is the most limiting factor for C dynamics. Thus the warming effect on soil water content, together with the general decrease in precipitation predicted by climate models for dryland areas, may result in situations of extremely limited soil water supply. The impact of low water availability in dryland areas on soil C is shown in the work of Li et al. (2015), who estimated a loss of 0.46 Pg C in Central Asia drylands during the 10-year drought period from 1998 to 2008, possibly related to extended La Niña episodes. Decreases in soil moisture limit microbial activity and, thus, soil organic matter decomposition (Skopp et al. 1990). Indeed, acceleration of microbial activity as a response of warming might be offset by exceptionally limited soil moisture (Almagro et al. 2009). However, the adoption of certain management practices could ameliorate this situation by increasing soil water available for crop growth and microbial activity. One main strategy would be tillage systems and in particular decreasing soil tillage intensity, since it has been identified as a promising management strategy to increase soil water content in dryland systems (Cantero-Martínez et al. 2007). Under a climate change scenario, the complete elimination of tillage through the adoption of no-tillage could help to maintain or even to increase crop growth and, thus, C inputs into the soil. But, it is important to consider that depending on the warming and drought extent, the adoption of this technique could stimulate soil C losses, due to an acceleration of soil microbial activity, which may not be compensated by the increase in C inputs. This last situation would imply C losses under no-tillage systems. Simulation studies in dryland systems under different climate change scenarios predicted future increases in SOC under no-tillage (Álvaro-Fuentes and Paustian 2011). Obviously, more experimental data is needed to determine the effect of no-tillage and other management practices on soil C changes under climate change conditions.

2.5. Social and economic barriers and opportunities

Drylands sustain over 2 billion people and contribute to climate change mitigation (Neely et al. 2009). Environmental and social co-benefits resulting from increased soil C sequestration in drylands can increase agroecosystems resilience and decrease social vulnerability to disasters and climate variability (Lipper et al. 2010). Past investments in drylands focused on improved land productivity by expansion of irrigated areas. This approach is unsustainable in most agricultural areas. Furthermore, dryland policies need to consider poverty reduction and environmental benefits.

2.5.1. Improved management viewed as an externality

Soils in dryland areas have potential social and economic benefits to improve sustainability of agricultural systems, environmental restoration, and poverty alleviation. Evidence for the benefits for increasing dryland C is clear at the local (i.e. increased crop productivity), regional (i.e. enhanced agricultural sustainability) and global levels (i.e. mitigation of climate change). As a consequence, the resulting benefits of the actions of farmers may produce positive externalities on other stakeholders and may take effect in the present or future.

The presence of externalities implies the need for policy interventions to ensure that improved C management is produced at the social optimum. Policy may provide incentives to farmers to produce this social optimum through various mechanisms, such as improved technical knowledge at the farmer level or improved carbon trading schemes. Understanding uncertainty and how to evaluate the future benefits is a major challenge and includes defining the value that we give future goods.

2.5.2. Measures at farmer level and policy support

At the farmer level, the main barriers are the initial investments. These investments are difficult to quantify, ranging from additional machinery to improved knowledge. The expected benefits at the farmer level may be insufficient to compensate farmers for the direct initial costs. Therefore policy interventions are necessary. In regions where agriculture is heavily supported by policy (i.e. Europe, USA, Australia) most studies conclude that subsidies are necessary. In regions where farmers do not receive direct support, substantial funds from development organizations or C investors will be necessary in order to make soil C sequestration projects in dryland small-scale farming systems a reality (Neely et al. 2009).

In the short term, changes in management are implemented first by the most interested, motivated and innovative farmers, that are often the ones that have other social and economic advantages. Marginal farmers are usually reluctant to participate in innovative programs and need different type of policy support. In the long term, the potential benefit of management practices that enhance C sequestration can be reversed as soon as they are abandoned. This might occur either as a consequence of natural hazards (such as a large drought), decreased policy support, or perspective of larger profits with another management alternative.

The success of a long-term and large-scale C sequestration program in drylands relies on the implementation by a large number of farmers. Top-down policy programs may only be successful if they provide financial incentives for implementation. At the same time, a program may build on already existing local and/or regional initiatives by farmers associations, for example. This would ensure that the measures proposed are supported by a large number of individuals.

2.5.3. Mainstreaming global development policies with C sequestration in drylands

The process of land degradation in drylands also means that C stored in these ecosystems will be added to the atmosphere as greenhouse gas emissions. It is also clear that extensive land degradation in drylands may contribute to poverty increase in many regions. A purely carbon-market approach is unlikely to be successful for drylands since the approach needs to consider other aspects such as sustainable development and poverty alleviation. Then, the adoption of carbon management strategies, which aims also at providing important co-benefits (e.g. climate change adaptation, biodiversity, plant nutrition, etc.) will gain more attraction in the mid- and long-term perspective. Sustainable carbon sequestration policies must act locally at the scale of the small shareholder or village, and focus on the ecosystem services rather than on C sequestration solely (Marks et al. 2009).

Therefore dryland C improvement policies are included into global development policies. This process is often referred to as mainstreaming, which is funded under other policies and could also be used to fund C sequestration programs in drylands. For example, the Convention to Combat Desertification (CCD) and the UN Framework Convention on Climate Change (UNFCCC) share the goal of improved management of C in drylands and poverty alleviation. As a consequence, there is a range of global policy mechanisms to promote dryland C storage for alleviation of poverty in least developed countries, such as the UN Global Mechanism program and the GEF land degradation focal area or the GEF Adaptation Fund (FAO 2004).

A key element of soil rehabilitation in drylands is the restoration of organic matter which has been widely depleted due to tillage, overgrazing and deforestation (see preceding sections). The Clean Development Mechanism of the Kyoto Protocol does not include the possibility of payments for C sequestration in soils. However, other markets in carbon are being developed, which could enable developing countries to benefit from carbon trading for soil organic matter (Lipper et al. 2010).

3. Conclusions

Dryland areas comprise about 41% of the Earth surface and sustain over 38% of the world's human population. A meaningful fraction of C in dryland soils has been lost as a consequence of inadequate management practices and land use decisions. Global warming will exacerbate the current scarcity of water that most dryland areas face, thus adding great challenges for agricultural production and social development. However, with proper decisions soils in dryland areas have a large potential to sequester C and will result in positive regional and global externalities.

Over the next decade, research on C management in dryland areas should focus on proper agricultural and livestock management practices that maximize C storage in soils taking into account their entire C footprint. Raising CO_2 levels and concomitant warming could also lead to heat stress that could offset the potential gain in SOC stocks expected under CO_2 enrichment conditions. Precipitation capture, water retention in the soil and crop water use efficiency need to be maximized to guarantee an adequate soil cover and reduce soil erosion susceptibility. A range of agronomic practices such as crop residue management, soil management and fertilization, adequate design of cropping systems and the availability of adapted plant material can help to increase soil C sequestration in water-limited environments. Livestock integration in dryland systems must be optimized to couple the C and N cycles and to take profit of the greater residence time of the C sequestered at soil depth. Future research should focus on the feedbacks between soil biodiversity and C cycling in order to enhance ecosystem services. Moreover, the areas of study must be up-scaled in order to better represent complex landscape processes affecting C sequestration and to improve the comprehension of the interactive effects of management and global warming on C cycling in soils. Policy support should generate possibilities to strengthen farmers' own strategies to deal with uncertainty while providing the necessary incentives to encourage successful C management pathways including an improved knowledge at the farmer level and strengthen the linkage between environmental and social sciences. The objective of sequestering a significant amount of C in dryland soils is attainable and will result in social and environmental benefits.

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Figure captions

Figure 1 A semiarid dryland agricultural system in the Ebro valley (NE Spain): a tillage and fertilization experiment was established in 2010 in a commercial 4-yr no-tilled field devoted to winter cereal production. The impact of a single pass of disk plough (15 cm depth) before sowing (plots of the right) and of the maintenance of no-tillage (plots of the left) on crop performance is shown.

Figure 2 Livestock use of stubble and straw from winter cereals and forage grazed from fallows is a common feature of large dryland regions such as the Mediterranean basin. The activity contributes to maintain a mosaic of cultivated and natural areas enhancing ecosystem services. If properly managed, livestock integration in dryland areas contributes to the increase in soil organic carbon contents.

Figure 3 Approach to evaluate research needs for optimizing C management in dryland agroecosystems.

Figure 1

Figure 2

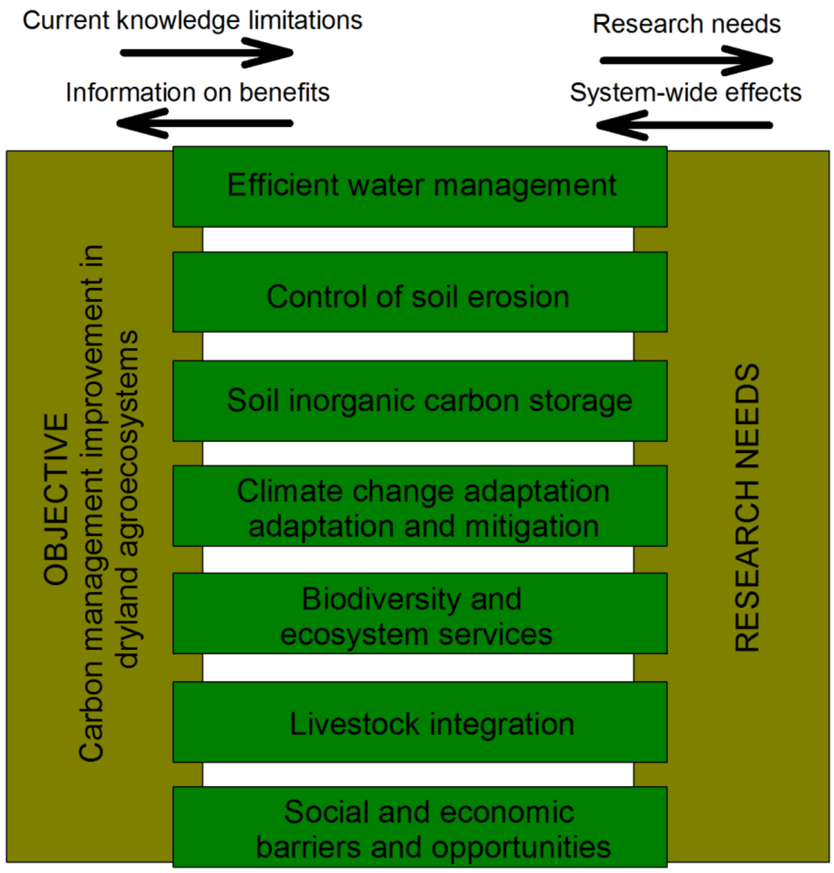
Current knowledge limitations
Information on benefits
Research needs
System-wide effects
OBJECTIVE
Carbon management improvement in dryland agroecosystems
Efficient water management
Control of soil erosion
Soil inorganic carbon storage
Climate change adaptation adaptation and mitigation
Biodiversity and ecosystem services
Livestock integration
Social and economic barriers and opportunities
RESEARCH NEEDS

Figure 3