Soil organic matter in dryland ecosystems

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Abstract

Drylands are regions with low rainfall, high temperatures and very high evapotranspiration, as well as limited plant biomass production. Covering over 45% of the Earth's land surface and being inhabited by more than 35% of the world population, drylands are of paramount importance for global sustainability. Among the many factors that influence the ability of drylands to provide essential ecosystem services, soil organic matter deserves special attention. This chapter provides an overview of the quantity of soil organic matter in dryland ecosystems, the main factors regulating its formation and conservation, its vulnerability to ongoing global changes, and existing options to preserve or even enhance this essential natural resource.

Keywords: Aridity; C cycle; C sequestration; Climate change; Drylands; Global Change; Grazing; Organic amendments; Soil organic matter; Tillage.

1. Introduction

Drylands are loosely defined as regions that receive intense solar radiation and relatively low average precipitation, resulting in a water deficit that limits net primary production (Safriel et al., 2005). More precisely, the United Nations Environment Program (UNEP) defines drylands as terrestrial areas where the aridity index (AI), or the ratio of total annual precipitation to potential evapotranspiration (evaporation from soil and transpiration by plants), is less than 0.65 (UNEP, 1997). Based on the AI, drylands are further divided into hyperarid (AI less than 0.05), arid (AI within the range from 0.05 to 0.2), semiarid (AI from 0.2 to 0.5), and dry subhumid regions (AI

from 0.5 to 0.65). The United Nations Convention to Combat Desertification (UNCCD) uses this classification, but addresses specifically only arid, semiarid, and dry subhumid areas as drylands for its purposes, excluding hyperarid areas (UNCCD, 1994). Instead of using the AI, the United Nations Food and Agriculture Organization (FAO) developed an index termed “length of growing period” (LGP) to identify the crop cultivation potential of land. The LGP, referring to the number of days in a year when temperature and soil moisture allow plant growth, is also used to classify general moisture regimes (IIASA/FAO, 2012). Areas with an LGP of 0, less than 60, from 60 to 119, and from 120 to 179 days are classified as hyperarid, arid, dry semiarid, and moist semiarid, respectively, which together correspond closely to the areas defined as drylands in the World Atlas of Desertification (FAO, 1993, 2004; IIASA/FAO, 2012).

Dryland ecosystems are of paramount significance both in terms of their extension and the human population they support. In particular, drylands cover over 45% of the Earth’s land surface, or about 6.7 billion hectares (Právělie, 2016), more than 70% of which is occupied by developing countries (FAO, 2004; Safriel et al., 2005; UN Environmental Management Group, 2011; Právělie, 2016) (Figure 1). Hyperarid, arid, semiarid, and dry subhumid regions account for about 13, 31, 36, and 20% of this area, respectively (Právělie, 2016). Most drylands can be ascribed to four biomes of decreasing aridity: deserts, grasslands, Mediterranean scrublands, and woodlands (Safriel et al., 2005). Of the global dryland area, approximately 65% is used as rangelands and 25% as cropland (Safriel et al., 2005). More than 2 billion people, or more than one third of the human population, live in drylands, and about 90% of these live in developing countries (Safriel et al., 2005; Middleton et al., 2011).

Soils make up an essential part of drylands, as in any other terrestrial ecosystem, as they constitute the medium for plant growth, provide a habitat for a wide range of organisms, regulate

70 water fluxes, and store C. The most common soils in drylands are Entisols (2.3 billion hectares)
71 and Aridisols (2.1 billion hectares), followed by Mollisols, Alfisols, and Vertisols (FAO, 2004).
72 Compared to humid-region soils, those in drylands are commonly characterized by low organic
73 matter and nutrient contents, frequent water stress, coarse to medium texture, moderate to high
74 base saturation, slightly acid to alkaline surface pH, accumulation of calcium carbonate, shallow
75 to moderate profile development, and low biological activity (Dregne, 1976; Skujins, 1991).
76 These properties result in inherently low fertility (Lal, 2004a).

77 Among the many factors that impact the quality and fertility of soils in drylands and their
78 ability to provide essential ecosystems services, soil organic matter deserves special attention.
79 This is because soil organic matter in drylands constitutes a huge global C reservoir and
80 influences almost all physical, chemical, and biological soil properties and processes in these
81 ecosystems (Lal, 2004a, 2009). Among other benefits, soil organic matter promotes the
82 formation of aggregates with mineral particles, which improves soil structure, porosity, and
83 moisture holding capacity, thus reducing the severity of water scarcity and protecting soils from
84 erosion and compaction. Also, soil organic matter contributes markedly to the soil acid-base
85 buffering capacity and represents an important source of plant nutrients. Furthermore, soil
86 biological diversity and activity are also strongly related to soil organic matter as a major source
87 of food for a variety of organisms (e.g., Paul et al., 1997; Bot and Benites, 2005; Paul, 2016).

88 This chapter provides an overview of the amount of soil organic matter stocks stored in
89 dryland ecosystems. It also explores the processes and factors affecting the balance between
90 accumulation and loss of soil organic matter, its vulnerability to global change, and the main
91 options available for its preservation and enhancement.

93

94 **2. Soil organic matter stocks in drylands**

95

96 Soil organic matter represents one of the largest active C reservoirs on Earth and thus plays a
97 significant role in the global C cycle with implications for the mitigation or exacerbation of
98 climate change (Batjes, 1996; Jobbágy and Jackson, 2000; Schmidt et al., 2011; IPCC, 2013).
99 In particular, while the atmosphere and global vegetation are estimated to contain 829 and 520
100 Pg of C, respectively (IPCC, 2013), global soil organic C mass is thought to be 1301 Pg within
101 the top 1 m, and about 2800 Pg when deeper soil layers are considered. This estimation is based
102 on the Harmonized World Soil Database (HWSD) v.1.2 (FAO/IIASA/ISRIC/ISSCAS/JRC,
103 2012; Köchy et al., 2015), corrected for the latest northern permafrost region inventory, which
104 estimates that permafrost soils store 472 Pg of organic C in the top 1 m, 1035 Pg in the top 3 m,
105 and 1330-1580 Pg in the known pool of terrestrial permafrost (Hugelius et al., 2014; Schuur et
106 al., 2015).

107 Globally, the largest soil organic C concentrations and stocks are in wetlands and peatlands,
108 generally in permafrost and tropical regions (Köchy et al., 2015). Soil organic C concentrations
109 in drylands are markedly smaller, often less than 0.5% of the soil mass, resulting in typical
110 densities of 0 to 15 kg m⁻² (Lal 2004a). However, due to the vast extension of dryland
111 ecosystems, the total amount of organic C stored in dryland soils is significant for the global C
112 cycle (Lal, 2004a). In particular, dryland soils are estimated to contain 431 Pg of organic C
113 (Safriel et al., 2005), which represents about 33% of the global soil organic C reserves in the
114 uppermost 1 m. Hyperarid and arid soils hold 113 Pg, whereas semiarid and dry subhumid soils
115 store 318 Pg. In addition to organic C, dryland soils are estimated to hold 916 Pg of inorganic C,

mainly as calcium carbonate, which accounts for 97% of the inorganic C stocks in the world's soils. Hyperarid and arid soils contain 732 Pg while semiarid and dry subhumid soils have 184 Pg (Safriel et al., 2005) (Figure 2).

As in the rest of terrestrial ecosystems, knowledge of the total soil organic C stock and its spatial distribution in drylands is essential to understand their vulnerability to global change and to develop strategies to recover, preserve, or enhance the services provided by their ecosystems. The current estimates of organic C stocks in dryland soils outlined above are still highly uncertain probably not accurate, being typically derived from soil databases that may underrepresent some regions (Köchy et al., 2015). Moreover, soil profile data are not collected with a consistent methodology and generally do not include soil layers deeper than 1 m (Darrouzet-Nardi, 2013). Nonetheless, most soil C in drylands is believed to reside in topsoil (Ciais et al., 2011), making it highly vulnerable to climate change and erosion process.

3. Natural factors affecting soil organic matter in drylands

Organic matter content in soils results from the balance between inputs, mainly from plant production, and losses, which occur mainly through *in situ* decomposition and respiration, but also through leaching, volatilization, erosion, and burning (Schulze and Freibauer, 2005). According to the latest concepts, plant material entering the soil is continuously processed by the decomposer community, producing fragments of smaller size with a concomitant surge in microbial byproducts (Plaza et al., 2013; Courtier-Murias et al., 2013; Lehmann and Kleber, 2015) (Figures 3). Soil organic matter decomposition is modulated by a number of

interconnected environmental, physical, chemical, and biological controls, such as temperature, moisture, activity and composition of microbial community, and organomineral interactions (Sollins et al., 1996; Six et al., 2002; Von Lützow et al., 2006; Schmidt et al., 2011). Freezing temperatures and high moisture content (low O₂) help preserve soil organic matter (climatic and environmental protection), as well as its association with aggregates (physical protection) and mineral surfaces (chemical protection) (Torn, 1997, 2009; Trumbore, 2009; Plaza et al., 2012, 2013) (Figure 4). During the degradation of plant material, organic substances are accumulated and protected from decomposition within aggregates and on mineral surfaces, limiting the access and action of decomposers (Golchin et al., 1994; Plaza et al., 2012, 2013; Courtier-Murias et al., 2013; Lehmann and Kleber, 2015).

More generally, according to the classic work by Hans Jenny (1941), soil organic matter stocks in drylands, like any other soil property in a given ecosystem, is a function of a number of soil forming factors, including climate, organisms (vegetation and soil biota), parent material, relief, and time. The main distinctive soil forming factors that drive soil organic matter storage in drylands are climate and vegetation. These state factors govern the balance of soil organic matter by influencing both the quantity and quality of plant input and the rate of decomposition. They also strongly influence the weathering and reactivity of soil minerals (thus affecting protective organomineral interactions), the structure and metabolic activity of the microbial community, and erosion as well as fire processes (Morgan, 2005; Lehmann et al., 2014).

Climate is the dominant factor explaining the low organic matter contents in dryland soils. By definition, drylands not only receive low precipitation, which limits plant production and organic matter inputs into the soil, but also have high temperatures, which encourage decomposition. In general, all else equal, regions with higher aridity tend to have smaller soil

organic matter stocks than do wetter regions. Furthermore, the rainfall regime of many dryland regions is characterized by events of short duration and high intensity (Nicholson, 2011), which favors soil organic matter losses by runoff and erosion (Martínez-Mena et al., 2002). Also, the low precipitation and high temperature increase the risk of wildfires in drylands (Middelton and Sternberg, 2013). Burning diminishes soil organic matter stocks over the short term; however, over the long term, wildfires might cause opposite effects because of the formation of organic compounds highly resistant to decomposition (Johnson and Curtis, 2001).

Vegetation affects soil organic matter stocks in drylands in several ways. Because of climate, plant productivity in drylands is low and therefore so is the rate of plant C inputs into the soil. However, the low water and nutrient availability generally favor other plant traits that help slow down decomposition and build up soil organic matter, such as poor-quality litter and a high root-to-shoot ratio that favors organic C allocation and protection deep into the soil (De Deyn et al., 2008). The spatial organization of vegetation, which organizes into discrete plant patches surrounded by areas devoid of vascular vegetation (Berdugo et al., 2017), is also relevant to soil organic matter stocks, particularly to their spatial distribution within dryland landscapes (Figure 5). The length of the growing season, which is highly variable in drylands, is also thought to have a significant influence on soil organic matter content (Lal, 2004a).

In addition to vegetation, biological soil crusts are a major biotic factor in organic C fixation in dryland soils. Biological soil crusts, or biocrusts, are communities of organisms dominated by lichens, mosses, bacteria, and fungi, inhabiting the soil surface (Belnap and Lange, 2003) (Figure 6). These communities have been found to be able to take up significant amounts of C and N from the atmosphere by photosynthesis and N fixation (Elbert et al., 2012), thus being an important pool and source of organic inputs into the soil and having a major role in soil

respiration in dryland ecosystems (Castillo-Monroy et al., 2011). Elbert et al. (2009) estimated the net uptake of C by biological soil crusts in drylands at $16 \text{ g m}^{-2} \text{ yr}^{-1}$, which corresponds to a total of approximately 1.0 Pg yr^{-1} for the global dryland area. Biocrusts are also known to protect soil from wind and water erosion, and thus from C loss, by strengthening soil structure through the interaction with mineral particles and formation of aggregates (Eldridge and Leys, 2003; Bowker, 2007; Belnap, 2013).

The abundance and strength of the organo-mineral interactions, and thus the physical and chemical protection of soil organic matter from decomposition, are thought to be lower in dryland soils than in soils of humid regions (Kleber et al., 2015). This is because water largely controls the weathering and formation of reactive sites on minerals needed to create organo-mineral associations. While wet conditions promote plant production, low pH conditions, soil weathering, and the formation of strong inner-sphere bonds between organic matter and mineral surfaces, energetically weaker outer-sphere complexation and H-bonds between organic matter and minerals are thought to dominate in dryland soils because of the lower moisture (Kleber et al., 2015).

4. Management activities affecting soil organic matter in drylands

Effects of grazing on soil organic matter in drylands

Livestock grazing is the main land use type in dryland regions and the primary livelihood for many of their inhabitants (Lal, 2004a). Grazing is known to largely affect the structure,

composition, and functioning of dryland ecosystems (Eldridge et al., 2016; Maestre et al. 2016; Gaitán et al., 2017), including their capacity to fix and store soil organic matter. Changes in soil organic matter stocks with grazing is mainly a function of its effects on net primary production (Piñeiro et al., 2010). Even though livestock may return a large proportion of nutrients and organic matter in the form of dung and urine (Haynes and Williams, 1999), this effect is likely not strong enough at a large scale to reinforce soil fertility in low-productive ecosystems (Bardgett and Wardle, 2003), such as drylands, and aboveground plant biomass is often reduced significantly by grazing. Using a large database from Australian rangelands, Eldridge et al. (2016) recently reported reductions of plant biomass by 40% and plant and litter cover by 25% after grazing, with more profound effects in drier environments and noticeable effects even at low grazing intensities. Using the same large set of records, Maestre et al. (2016) found that grazing, even at low intensity, negatively affected the C content and reported average reductions of approximately 8%. In addition to a decrease in aboveground plant inputs into the soil, livestock often increases soil compaction through trampling, thereby reducing water infiltration and availability to plants while encouraging runoff and soil erosion, thus further contributing to soil organic matter losses (Piñeiro et al., 2010; Eldridge et al., 2016). Intensive grazing has also been found to adversely affect soil organic matter stocks by reducing the cover, biomass, and diversity of the biological soil crusts through trampling and burial (Warren and Eldridge, 2003; Thomas, 2012). Increases in C inputs into the soil from greater root biomass due to changes in plant C allocation or species abundance, on the other hand, have been suggested to partially offset soil organic matter loss in drylands (Piñeiro et al., 2010).

Effects of cultivation on soil organic matter in drylands

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232 Cultivation is the second most widespread use of drylands (Safriel et al., 2005). The
233 conversion of natural ecosystems to croplands is well known to cause significant losses of soil
234 organic matter (Matson et al., 1997), and has been responsible for a large proportion of the
235 global anthropogenic CO₂ emissions (Lal, 2004b; IPCC, 2013). Meta-analysis results estimate
236 global soil C stock losses at 42% when converting from native forest to crop and at 59% from
237 pasture to crop (Guo and Gifford, 2002). As throughout the rest of the globe, the expansion and
238 intensification of cultivation in drylands have led to a substantial depletion of soil organic matter
239 stocks, concomitantly with more severe nutrient depletion and soil erosion. For example, it has
240 been estimated that cropping of Australian semiarid regions has resulted in long-term losses of
241 soil organic matter often exceeding 60% of the initial content (Dalal and Chan, 2001).

242 In general, the loss of soil organic matter with conversion to permanent agriculture can be
243 attributed to decreased input rates of plant-derived organic matter and increased decomposition
244 rates. Agricultural crops are harvested and typically leave less organic matter belowground than
245 does native vegetation (Post and Kwon, 2000). The rate and amount of organic matter loss
246 depend not only on natural factors, such as climate and soil properties, but also on the specific
247 cropping system. Farming methods that have particularly exacerbated the loss of soil organic
248 matter in drylands include the use of conventional intensive tillage and removal of crop residues
249 (McLeod et al., 2013).

250 Conventional intensive soil tillage, used for weed control and seedbed preparation, reduces
251 soil organic matter stocks by increasing soil aeration and facilitating the access of decomposers
252 to organic substrates, thus accelerating decomposition rates (I. Haller, personal communication)
253 (Figure 7). More specifically, soil tillage is thought to increase macroaggregate turnover, thereby

inhibiting the formation of microaggregates within macroaggregates in which particulate organic matter is stabilized over the long term (Six et al., 1998, 1999, 2000). It has also been proposed that the increased macroaggregate turnover may inhibit not only microaggregate formation and thereby the physical protection of crop-derived particulate organic matter, but also the interaction between mineral particles and microbial material that results in the formation of very stable organo-mineral complexes (Plaza et al., 2013). By disturbing the soil structure, tillage also makes soil organic matter more vulnerable to erosion losses.

5. Vulnerability of soil organic matter in drylands to global change

Humanity has a dominant and growing influence on the structure and functioning of all ecosystems on Earth (Vitousek et al., 1997). Because of their inherently low fertility and vegetation cover, dryland ecosystems are particularly vulnerable to human activities and global environmental change drivers, including global warming, invasion of alien species, and increased N deposition, which have profound social and economic implications (Reynolds et al., 2007).

According to the latest data from the Population Division of the UN Department of Economic and Social Affairs of the United Nation Secretariat (UN-DESA-PD, 2015), the world population continues to grow and is projected to reach 8.5 billion by 2030, 9.7 billion in 2050 and 11.2 billion in 2100. Most of the projected increase will occur in developing countries and in dryland regions. A rapidly increasing population will demand more food and livestock production (Thornton et al., 2010; Foley et al., 2011; Tilman et al., 2011). This will put more pressure on drylands, as cropland use intensifies and pastures are expanded, negatively affecting

soil organic matter stocks as described above. In fact, population growth is arguably the most critical challenge facing drylands to avoid further degradation and desertification.

Vulnerability of soil organic matter to climate change in drylands

Continued anthropogenic emissions of greenhouse gases, particularly CO₂, will spur further rises in global mean temperatures over the 21st century (IPCC, 2013). By the end of this century, dryland area is projected to increase by 23% under the current path of CO₂ emissions and global warming (Representative Concentration Pathway RCP8.5) and by 11% under a scenario of intermediate emissions in which total radiative forcing is stabilized shortly after 2100 (RCP4.5), relative to the 1961-1990 baseline (Huang et al., 2016). In other words, by 2100, 56% (RCP8.4) or 50% (RCP4.5) of the total land surface area is expected to be covered by drylands (Huang et al., 2016). Dryland expansion and humid region shrinkage will reduce global plant production, which will in turn diminish global soil organic matter stocks and augment CO₂ in the atmosphere.

In drylands, temperatures are expected to rise more than in humid regions, with 4 °C increases projected in some areas (IPCC, 2013). Precipitation patterns will be altered, extreme events such as droughts and heat waves will become more frequent, and the degree of aridity experienced by drylands will increase (IPCC, 2013; Fu and Feng, 2014; Cook et al., 2015; Huang et al., 2016). All these interwoven factors are expected to reduce the overall plant cover (Delgado-Baquerizo et al., 2013) and the abundance and diversity of soil bacteria and fungi (Maestre et al., 2015), exerting negative consequences on C cycling and storage (Delgado-Baquerizo et al., 2016). Furthermore, the expected increases in aridity with climate change are

expected to negatively affect the abundance, composition, and diversity of biocrusts, thereby compromising their organic C sink capacity (Maestre et al., 2013; Ferrenberg et al., 2015).

Vulnerability of soil organic matter to vegetation shift in drylands

Vegetation shifts, and particularly the invasion of native grasslands by woody species (shrub encroachment) or exotic grasses, is a widespread phenomenon long documented in many dryland areas (Archer, 1994; Van Auken, 2000; Biedenbender et al., 2004; Maestre et al., 2009; Brazier et al., 2014). A number of studies suggest that plant invasions of grasslands in dry environments tend to boost the soil organic C content in the absence of human and natural disturbance (Jackson et al., 2002; Barger et al., 2011), while a global meta-analysis reported significant increases in total and organic soil C with shrub encroachment (Eldridge et al., 2011). In North American drylands, trends of soil organic C accumulation with woody plant encroachment were found not to be coupled to aboveground net primary production, which tended to decrease in arid regions and to increase in semiarid and subhumid regions (Barger et al., 2011). Changes in vegetation have often been reported to be accompanied by greater run-off and erosion, which may significantly offset gains in plant C inputs and lead to net soil organic matter losses (Turnbull et al., 2010; Brazier et al., 2014; Puttock et al., 2014). For example, recent studies across a shrub encroachment gradient in the southwestern United States indicated that soil organic C stored beneath the vegetation increased relative to that of bare soil areas as the grass cover became sparser and shrubs became more established in the landscape (Brazier et al., 2014; Puttock et al., 2014). In this case, there was more heterogeneity of soil organic C with encroachment from a more homogeneous distribution under grass to a patchy distribution under shrubs; and this spatial

heterogeneity made soil and soil organic C more susceptible to erosion from areas between shrubs.

Vulnerability of soil organic matter to N deposition in drylands

Human activities, such as the burning of fossil fuels and the use of N fertilizers, have tremendously accelerated the N cycle and have significantly exacerbated the release of N into the atmosphere (Gruber and Galloway, 2008). This excess of N is not only contributing to global warming but is also resulting in heavier atmospheric deposition of N in terrestrial ecosystems worldwide (Gruber and Galloway, 2008). Plant growth in drylands, as in most ecosystems, is naturally N limited (Waldrop et al., 2004; Maestre et al., 2016; Ochoa-Hueso et al., 2016). The addition of N may thus enhance plant growth and organic matter inputs (Maestre et al., 2016; Ochoa-Hueso et al., 2016). Increased N deposition, however, can also favor fast-growing plant species at the expense of the slow-growing ones and thereby reduce native plant diversity (Bobbink et al., 2010; Lan and Bai, 2012). Other effects of N deposition that may alter the ability of drylands to store soil organic matter include soil acidification, stronger aluminum toxicity, and changes in soil microbial composition, biomass, and activity (Bardgett, 2005; Ochoa-Hueso et al., 2011). In general, N enrichment can stimulate cellulolysis and labile organic matter decomposition, while suppressing the ligninolytic activity of fungi (Fog, 1988). The structure and function of biocrusts are also reportedly affected by N deposition, the most significant responses being reductions in photosynthesis rates and shifts in cyanobacteria community (Ochoa-Hueso et al., 2013). Although the available data are scarce, all these potential operating factors identified in the literature suggest that the effects of N deposition for soil organic matter

storage in non-agricultural drylands may be highly context dependent. Soil organic C content declines along a gradient of N deposition in low-productivity semiarid Mediterranean areas (Ochoa-Hueso et al., 2013), while soils of other different ecosystems subject to elevated N additions appear to be soil organic matter sinks (Lloyd, 1999; Zak et al., 2016).

As a whole, ample evidence implies that the ongoing global environmental change will substantially affect the structure and function of dryland ecosystems, including their ability to store soil organic matter. It is important to note that global change phenomena occur simultaneously, and therefore their effects on ecosystems may lead to highly significant interactions between them (Bardgett, 2005). However, little is known about the magnitude of these interactions and how they might affect soil organic matter stocks in drylands, albeit there is evidence of biological feedbacks between climate change and changes in biocrust communities can alter them (Maestre et al., 2013).

6. Practices to restore, preserve, and augment soil organic matter in drylands

Soil organic matter is universally recognized to be a central factor enabling ecosystems to provide key services needed for global sustainability, including food and wood production, CO₂ fixation and soil stability (Haygarth and Ritz, 2009; see chapter by Masciandaro et al. in this book). Yet dryland soils have been a historic source of CO₂ emissions to the atmosphere and have lost massive amounts of their native soil organic matter contents due to land management practices, degradation, and desertification. In the dryland ecosystems of West Asia and North Africa alone, which represent roughly 25% of the global dryland area, the historic organic C loss

is thought to have been between 6 and 12 Pg (Lal, 2001a). Desertification has caused the loss of 20 to 30 Pg of C globally (Lal, 2001b, 2004a). The magnitude of this loss shows the potential and the urgent need to develop strategies to restore, preserve, or increase soil organic matter in drylands, as these strategies may help bolster productivity, prevent erosion, and degradation, and mitigate climate change, with substantial local and global societal, economic, and environmental benefits. Soil organic matter contents can be built up through management practices that increase organic inputs into the soil and decrease organic matter decomposition, erosion, and leaching, or both (Lal, 2004a, 2009). These practices include cropland conversion to grassland and afforestation, improved grazing regime, and improved farming management, such as using organic amendments, mineral N fertilization, cover crops, crop rotations, shifting from conventional tillage to conservation tillage practices, and improving water use efficiency (Lal, 2004a, 2004c) (Figure 8). The potential of increasing soil C in drylands is estimated to be about 60-70% of the historic C loss in 25 to 50 years (Lal, 2001a, 2001b, 2004a).

Cropland conversion to grasslands and afforestation

Land conversion from natural to agricultural ecosystems has been the main cause for the historic depletion of soil organic matter stocks (Lal, 2004c). There is abundant evidence that, in drylands, shifts from crop cultivation to grass or tree based systems, including conversion to pasture and agroforestry production systems, can help in the recovery, at least partially, of the historic soil organic matter pools (Lal, 2001a, 2001b, 2004a). In a semiarid watershed in northern Ethiopia, for example, compared to rainfed crop production, soil organic C stocks to 30 cm increased by 52% with based fruit production, 60% with agroforestry based crop production,

143% with silvopasture, and 227% with open communal pasture after 50 years after land use conversion (Gelaw et al., 2014). In the Highveld of semiarid South Africa, with the use of a chronosequence and modeling approach, soil organic C stocks to 20 cm deep were estimated to increase after cropland conversion to permanent grassland for use as pasture, reaching maximum values after 10 to 95 years, which were 30% to 94% higher than in adjacent arable soils, but 28% to 53% lower than in native grasslands (Preger et al., 2010). These latter results strongly suggest that, in native grassland soil, organic matter loss cannot be entirely recovered after land use change.

Soil organic matter accrual after afforestation depends on a number of factors, such as previous land use, plantation age, soil properties, tree species, and climate (Langanière et al., 2010; Berthrong et al., 2012; Li et al., 2012). In the long term, C accumulation is likely to be greater in croplands than in grasslands, and in pastures than in natural grasslands, with broadleaf than with coniferous species, in clay-rich than in coarse-textured soils, and in warmer and dryer than in cooler and wetter regions (Langanière et al., 2010; Berthrong et al., 2012; Li et al., 2012). For instance, in the semiarid Loess Plateau of China, cropland afforestation increased soil organic C stocks after an initial depletion period, with higher relative gain as precipitation declined (Chang et al., 2014). In many dryland ecosystems, the impacts of afforestation on soil properties and C sequestration also highly depend on the planting technique employed. In Southeastern Spain, plantations of tree species such as *Pinus halepensis* frequently show enhanced runoff and soil losses when compared to adjacent natural shrublands, as well as limited improvement in most physico-chemical properties, which rarely reach the values shown by natural shrublands even 40 years after planting (Maestre and Cortina, 2004).

Improved grazing management

Intensive livestock grazing has been shown to affect the structure, composition, and functions of dryland ecosystems, including their ability to store soil organic matter mainly by reducing plant cover, decreasing plant-derived organic inputs, and destroying biocrusts (Thomas, 2012; Eldrige et al., 2016). In general, improved practices aimed at lowering grazing intensity, alleviating soil disturbance, and lengthening soil and plant recovery time can help reduce soil organic matter loss and restore C sequestration capacity (Thomas, 2012). In semiarid grasslands of South Africa, soil organic C stocks to 20 cm in farms with rotational grazing were higher by a factor of 1.4 relative to areas with continuous grazing, commonly overstocked and with shorter soil and vegetation resting times (Kotzé et al., 2013). Destocking grazed Australian semiarid shrublands for two decades increased soil C stocks by about 6.5 Mg ha⁻¹ (Daryanto et al., 2013). Selective grazing can also reduce soil organic matter loss by promoting plant traits connected to slow C turnover (Bardgett and Wardle 2003; De Deyn et al., 2008). Controlled stubble grazing after crop harvest vs. removal of crop residues is another management option to reduce erosion and maintain or build up organic matter in dryland soils (FAO, 2004; Lal, 2004b; Blanco-Canqui et al., 2016; Stavi et al., 2016).

Improved management practices in croplands

Organic amendments. The application of organic amendments to the soil is a long-established management practice proven to be highly effective to replenish and augment soil organic matter in croplands (Ogle et al., 2005; Senesi et al., 2007; Paustian., 2014). Organic

amendments can build up soil organic matter both directly by adding organic C and indirectly by enhancing C inputs from crop residues through improving total plant biomass production (Fernández et al., 2009) (Figure 9). Traditional organic products that have been used for thousands of years include crop residues and animal manures. Organic amendments typically originate from waste materials subjected to appropriate treatments, such as composting. These treatments aim to destroy pathogens, eliminate phytotoxic substances, and stabilize the organic matter components, so that, once in soil, they decompose relatively slowly (Senesi et al., 2007). Long-term experiments have shown that the application of organic amendments, such as crop residues, manure, sewage sludge, and municipal solid waste compost, can increase soil organic matter contents by up to 90% (Diacono and Montemuro, 2010), and the gains in drylands can be long-lasting even after just a single application (Bastida et al., 2008).

Among the wide range of organic amendments available to farmers, biochar has recently emerged as a highly promising product for climate change mitigation. Biochar is made by the pyrolysis of biomass, and biochar C is believed to form very stable structures, which can be stored in soils for much longer than the biomass C from which it originates (Lehmann, 2007; Lehmann and Joseph, 2015) (Figure 10). The typically high pH values of biochar might be a limiting factor for its application in highly alkaline dryland soils. Nonetheless, under semiarid conditions biochar may also help stabilize organic C from other amendments co-applied to soil by promoting organo-mineral associations (Plaza et al., 2016) (Figure 11).

Mineral fertilization. The correct application of mineral fertilizers, particularly N fertilizers, increase crop production and plant-derived organic inputs into the soil. These inputs, however, may not be substantial enough to offset organic matter losses through mineralization in drylands, and current evidence suggests that it is particularly important to combine mineral fertilization

with other recommendable agricultural practices to efficiently increase soil organic C stocks. For example, repeated mineral fertilization for 26 years at an arid site in China did not affect the total soil organic matter contents (Hai et al., 2010). Similarly, in a 7-year field experiment in northwestern China under arid conditions, mineral fertilization had little effect on soil organic matter contents, but improved the performance of organic manure (Yang et al., 2016). Practices that retain crop residues in the field have been shown to benefit particularly from mineral fertilization for soil organic matter accrual (Naab et al., 2015).

Crop rotation. Compared to the practice of growing the same crop for many years on the same field (monoculture), crop rotation avoids the disproportional depletion of nutrients and controls weeds as well as pests, thereby improving above- and belowground biomass production. Growing different crops, especially crops that produce large amount of residues, can thus augment soil organic matter stocks by increasing organic matter inputs into the soil (West and Post, 2002; Ogle et al., 2005; Palm et al., 2014). For example, an experiment conducted at a semiarid site in the Argentine Pampa showed 3% more soil organic C stocks to 100 cm after 15 years under a soybean-maize alternation than under a soybean monoculture (Alvarez et al., 2014).

Cover crops. A cover crop, consisting of plants planted between cash crops to conserve and enrich the soil, can provide multiple benefits, such as preventing soil erosion, storing nutrients, suppressing weeds, controlling pests and diseases, and improving soil structure, among others. In a recent review, Blanco-Canqui et al. (2015) showed that cover crops can boost soil organic C stocks from 0.01 to 0.1 kg m⁻² yr⁻¹, depending on factors such as initial soil organic matter content, biomass production, and climate. In drylands, cover crops can build up soil organic matter by protecting soil from water and wind erosion and by incorporating extra plant residues

into the soil. However, cover crops may depress main crop yields in water-limited regions by reducing water availability (Blanco-Canqui et al., 2015), which may counterbalance to some extent the organic matter inputs. Nonetheless, a number of studies in drylands indicate that reducing or eliminating the fallow period by using cover crops, especially if combined with practices that enhance water storage such as no tillage, can significantly increase plant-derived organic matter inputs and slow erosion and decomposition rates, resulting in net gains in soil organic matter stocks (Blanco-Canqui et al., 2013; Paustian, 2014; Mitchell et al., 2017).

Conservation tillage. In contrast to intensive tillage, conservation practices that entail little or no soil disturbance can maintain and even build up surface soil organic matter by slowing macroaggregate turnover, encouraging microaggregate formation, physically protecting crop-derived particulate organic matter (Paustian et al., 2000; West and Post, 2002; Six et al., 2000, 2004; Plaza et al., 2013), and prompting the formation of primary organo-mineral complexes, particularly with microbial material (Plaza et al., 2013) (Figures 12 and 13). A recent meta-analysis using 46 peer-reviewed publications and 174 paired observations showed that tilled soils emitted 21% more CO₂ than untilled soils across different climates, crop types, and soil conditions, and the difference increased to 29% in sandy soils from arid climates with low soil organic C content and low soil moisture (Abdalla et al., 2016). In semiarid central Spain, 25 years of no-tillage, compared to chisel plowing, resulted in 16% more organic C in the uppermost 20 cm of soil, especially because of an enhanced interaction between mineral particles and microbial material that resulted in the formation of highly stable organo-mineral complexes (Plaza et al., 2013). In southern Texas, under subhumid conditions, no-tillage resulted in 64% greater soil organic matter stocks compared with the average pools resulting from different tillage treatments of increasing intensity after 16 years (Salinas-Garcia et al., 1997).

Efficient irrigation. Matching irrigation to crop water demands can raise yields, crop residue production, and plant-derived inputs into the soil, thus building up soil organic matter stocks (Zhang et al., 2015). Especially in areas where freshwater is scarce, the use of wastewater for agricultural irrigation is becoming steadily more important because of the increasing pressure on water resources (Lado and Ben-Hur, 2009). In addition to being a more sustainable practice of water management, wastewater irrigation has the additional benefit of enriching soil fertility by adding nutrients; nonetheless, the safe and efficient use of effluents requires a careful control of potential risks, such as soil salinization and contamination (Halliwell et al., 2001; Lado and Ben-Hur, 2009). At three sites in southeastern Spain, irrigation with wastewater for 20 to 40 years was found not to adversely affect soil properties and in fact increased soil organic matter stocks with respect to irrigation with freshwater (Morugan-Coronado et al., 2013). However, some studies have pointed out the risk of greater mineralization of native soil organic matter (positive priming effects) because of the addition of fresh dissolved organic matter (Lado and Ben-Hur, 2009).

7. Concluding remarks

As a whole, soil organic C concentrations in drylands are very low compared to those in other ecosystems. At the same time, drylands cover 45% of the Earth's land surface, and thus the total amount of organic C stored in drylands becomes a significant component of the global C cycle. Unfortunately, while other ecosystems traditionally considered as C sinks (e.g., peatlands) have been intensively investigated, estimates of organic C stocks in dryland soils are still very

poor and need to be improved strongly. This is extremely important in order to understand their vulnerability to climate change, as well as to develop strategies to recover, preserve, or enhance the ecosystem services provided, which are mostly affected by soil organic matter. Moreover, most drylands are located in poor or developing countries and host about one third of the global human population. As a consequence, further C losses from these soils may have long-term adverse impacts on soil quality and global food security and, in turn, on social unrest and political instability. In conclusion, soil organic matter plays a key role in soil conservation in several countries, and in drylands in particular, where it represents “the link” among global issues, including land degradation, food security, and climate change.

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References

553 Abdalla, K., Chivenge, P., Ciais, P., Chaplot, V., 2016. No-tillage lessens soil CO₂ emissions the
554 most under arid and sandy soil conditions: results from a meta-analysis. *Biogeosciences* 13,
555 3619-3633.

556 Alvarez, C., Alvarez, C.R., Costantini, A., Basanta, M., 2014. Carbon and nitrogen sequestration
557 in soils under different management in the semi-arid Pampa (Argentina). *Soil and Tillage*
558 *Research* 142, 25-31.

559 Archer, S., 1994. Woody plant encroachment into southwestern grasslands and savannas: rates,
560 patterns and proximate causes. In: Vavra, M., Laycock, W., Pieper, R. (Eds.), *Ecological*
561 *Implications of Livestock Herbivory in the West*, Society for Range Management, Denver,
562 CO, pp. 13-68.

563 Bardgett, R.D., 2005. *The Biology of Soil*. Oxford University Press, Oxford, UK.

564 Bardgett, R.D., Wardle, D.A., 2003. Herbivore-mediated linkages between aboveground and
565 belowground communities. *Ecology* 84, 2258-2268.

566 Barger, N.N., Archer, S.R., Campbell, J.L., Huang, C., Morton, J.A, Knapp, A.K., 2011. Woody
567 plant proliferation in North American drylands: A synthesis of impacts on ecosystem carbon
568 balance. *Journal of Geophysical Research Biogeosciences* 116, G00K07.

569 Bastida, F., Kandeler, E., Hernández, T., García, C., 2008. Long-term effect of municipal solid
570 waste amendment on microbial abundance and humus-associated enzyme activities under
571 semiarid conditions. *Microbial Ecology* 55, 651-661.

572 Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil*
573 *Science* 47, 151-163.

574 Belnap, J., 2013. Biological soil crusts and wind erosion. In: Belnap, J., Lange, O.L. (Eds.),
575 *Biological Soil Crusts: Structure, Function, and Management*. Springer, Berlin, Germany.

576 Belnap, J., Lange, O.L., 2013. Biological Soil Crusts: Structure, Function, and Management.
577 Springer, Berlin, Germany.

578 Berdugo, M., Kéfi, S., Soliveres, S., Maestre, F.T., 2017. Plant spatial patterns identify
579 alternative ecosystem multifunctionality states in global drylands. *Nature Ecology and*
580 *Evolution* 1, 0003.

581 Berthrong, S.T., Piñeiro, G., Jobbágy, E.G., Jackson, R.B., 2012. Soil C and N changes with
582 afforestation of grasslands across gradients of precipitation and plantation age. *Ecological*
583 *Applications* 22, 76-86.

584 Biedenbender, S.H., McClaren, M.P., Quade, J., Wertz, M.A., 2004. Landscape patterns of
585 vegetation change indicated by soil carbon isotope composition. *Geoderma* 119, 69-83.

586 Blanco-Canqui, H., Holman, J.D., Schlegel, A.J., Tatarko, J., Shaver, T.M., 2013. Replacing
587 fallow with cover crops in a semiarid soil: Effects on soil properties. *Soil Science Society of*
588 *America Journal* 77, 1026-1034.

589 Blanco-Canqui, H., Shaver, T.M., Lindquist, J.L., Shapiro, C.A., Elmore, R.W., Francis, C.A.,
590 Hergert, G.W., 2015. Cover crops and ecosystem services: insights from studies in temperate
591 soils. *Agronomy Journal* 107, 2449-2474.

592 Blanco-Canqui, H., Tatarko, J., Stalker, A.L., Shaver, T.M., Van Donk, S.J., 2016. Impacts of
593 corn residue grazing and baling on wind erosion potential in a semiarid environment. *Soil*
594 *Science Society of America Journal* 80, 1027-1037.

595 Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante,
596 M., Cinderby, S., Davidson, E., Dentener, F., Emmet, B., Erisman, J.-W., Fenn, M., Gilliam,
597 F., Nordin, A., Pardo, L., De Vries, W., 2010. Global assessment of nitrogen deposition
598 effects on terrestrial plant diversity: a synthesis. *Ecological Applications* 20, 30-59.

599 Bot, A., Benites, J., 2005. The Importance of Soil Organic Matter. Key to Drought-resistant Soil
600 and Sustained Food and Production. FAO Soils Bulletin 80. FAO, Rome, Italy.

601 Bowker, M.A., 2007. Biological soil crust rehabilitation in theory and practice: An
602 underexploited opportunity. *Restoration Ecology* 15, 13-23.

603 Brazier, R.E., Turnbull, L., Wainwright, J., Bol, R., 2014. Carbon loss by water erosion in
604 drylands: implications from a study of vegetation change in the south-west USA.
605 *Hydrological Processes* 28, 2212-2222.

606 Castillo-Monroy, A.P., Maestre, F.T., Rey, A., Soliveres, S., García-Palacios, P., 2011.
607 Biological soil crust microsites are the main contributor to soil respiration in a semiarid
608 ecosystem. *Ecosystems* 14, 835-847.

609 Chang, R., Jin, T., Lü, Y., Liu, G., Fu, B., 2014. Soil carbon and nitrogen changes following
610 afforestation of marginal cropland across a precipitation gradient in Loess Plateau of China.
611 *PLoS ONE* 9, e85426.

612 Ciais, P., Bombelli, A., Williams, M., Piao, S.L., Chave, J., Ryan, C.M., Henry, M., Brender, P.,
613 Valentini, R., 2011. The carbon balance of Africa: synthesis of recent research studies.
614 *Philosophical Transactions of the Royal Society A* 369, 2038-2057.

615 Cook, B.I., Ault, T.R., Smerdon, J.E., 2015. Unprecedented 21st century drought risk in the
616 American Southwest and Central Plains. *Science Advances* 1, e1400082.

617 Courtier-Murias, D., Simpson, A.J., Marzadori, C., Baldoni, G., Ciavatta, C., Fernández, J.M.,
618 López-de-Sá, E.G., Plaza, C., 2013. Unraveling the long-term stabilization mechanisms of
619 organic materials in soils by physical fractionation and NMR spectroscopy. *Agriculture,*
620 *Ecosystems and Environment* 171, 9-18.

621 Dalal, R.C., Chan, K.Y., 2001. Soil organic matter in rainfed cropping systems of the Australian

622 cereal belt. Australian Journal of Soil Research 39, 435-464.

623 Darrouzet-Nardi, A., 2013. How much carbon is stored in dryland soils?
624 <http://anthony.darrouzet-nardi.net/scienceblog/?p=1664> (accessed 10 October 2016).

625 Daryanto, S., Eldridge, D.J., Throop, H.L., 2013. Managing semi-arid woodlands for carbon
626 storage: grazing and shrub effects on above- and belowground carbon. Agriculture,
627 Ecosystems and Environment 169, 1-11.

628 De Deyn, G.B., Cornelissen, J.H.C., Bardgett, R.D., 2008. Plant functional traits and soil carbon
629 sequestration in contrasting biomes. Ecology letters 11, 516-531.

630 Delgado-Baquerizo, M., Maestre, F.T., Gallardo, A., Bowker, M.A., Wallenstein, M.D., Quero,
631 J.L., Ochoa, V., Gozalo, B., García-Gómez, M., Soliveres, S., García-Palacios, P., Berdugo,
632 M., Valencia, E., Escolar, C., Arredondo, T., Barraza-Zepeda, C., Bran, D., Carreira, J.A.,
633 Chaieb, M., Conceição, A.A., Derak, M., Eldridge, D.J., Escudero, A., Espinosa, C.I.,
634 Gaitán, J., Gatica, M.G., Gómez-González, S., Guzman, E., Gutiérrez, J.R., Florentino, A.,
635 Hepper, E., Hernández, R.M., Huber-Sannwald, E., Jankju, M., Liu, J., Mau, R.L., Miriti,
636 M., Monerris, J., Naseri, K., Noumi, Z., Polo, V., Prina, A., Pucheta, E., Ramírez, E.,
637 Ramírez-Collantes, D.A., Romão, R., Tighe, M., Torres, D., Torres-Díaz, C., Ungar, E.D.,
638 Val, J., Wamiti, W., Wang, D., Zaady, E., 2013. Decoupling of soil nutrient cycles as a
639 function of aridity in global drylands. Nature 502, 672-676.

640 Delgado-Baquerizo, M., Maestre, F.T., Reich, P.B., Jeffries, T.C., Gaitán, J.J., Encinar, D.,
641 Berdugo, M., Campbell, C.D., Singh, B.K., 2016. Microbial diversity drives
642 multifunctionality in terrestrial ecosystems. Nature Communications 7, 10541.

643 Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility.
644 A review. Agronomy for Sustainable Development 30, 401-422.

645 Dregne, H.E., 1976. Soils of Arid Regions. Developments in Soil Science 6. Elsevier,
646 Amsterdam, The Netherlands.

647 Elbert, W., Weber, B., Büdel, B., Andreae, M.O., Pöschl, U., 2009. Microbiotic crusts on soil,
648 rock and plants: neglected major players in the global cycles of carbon and nitrogen?
649 Biogeosciences Discussions 6, 6983-7015.

650 Elbert, W., Weber, B., Burrows, S., Steinkamp, J., Büdel, B., Andreae, M.O., Pöschl, U., 2012.
651 Contribution of cryptogamic covers to the global cycles of carbon and nitrogen. Nature
652 Geoscience 5, 459-462.

653 Eldridge, D.J., Leys, J.F., 2003. Exploring some relations between biological soil crusts, soil
654 aggregation and wind erosion. Journal of Arid Environments 53, 453-466.

655 Eldridge, D.J., Bowker, M.A., Maestre, F.T., Reynolds, J.F., Roger, E., Whitford, W.G., 2011.
656 Impacts of shrub encroachment on ecosystem structure and functioning: towards a global
657 synthesis. Ecology Letters 14, 709-722.

658 Eldridge, D.J., Poore, A.G.B., Ruiz-Colmenero, M., Letnic, M., Soliveres, S., 2016. Ecosystem
659 structure, function, and composition in rangelands are negatively affected by livestock
660 grazing. Ecological Applications 26, 1273-1283.

661 FAO, 1993. Land degradation in arid, semi-arid and dry sub-humid areas: rainfed and irrigated
662 lands, rangelands and woodlands. Text for FAO presentation at inter-governmental
663 negotiating committee for the preparation of a convention to combat desertification and
664 drought (INCD). FAO, Rome, Italy.

665 FAO, 2004. Carbon Sequestration in Dryland Soils. World Soil Resources Reports 102. FAO,
666 Rome, Italy.

667 FAO/IIASA/ISRIC/ISSCAS/JRC, 2012. Harmonized World Soil Database (version 1.2). FAO,

668 Rome, Italy and IIASA, Laxenburg, Austria.

669 Fernández, J.M., Plaza, C., García-Gil, J.C., Polo, A., 2009. Biochemical properties and barley
670 yield in a semiarid Mediterranean soil amended with two kinds of sewage sludge. *Applied*
671 *Soil Ecology* 42, 18-24.

672 Ferrenberg, S., Reed, S.C., Belnap, J., 2015. Climate change and physical disturbance cause
673 similar community shifts in biological soil crusts. *Proceedings of the National Academy of*
674 *Sciences of the United States of America* 112, 12116-12121.

675 Fog, K., 1988. The effect of added nitrogen on the rate of decomposition of organic matter.
676 *Biological Reviews of the Cambridge Philosophical Society* 63, 433-462.

677 Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller,
678 N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill,
679 J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks,
680 D.P.M., 2011. *Nature* 478, 337-342.

681 Fu, Q., Feng, S., 2014. Responses of terrestrial aridity to global warming. *Journal of Geophysical*
682 *Research Atmospheres* 119, 7863-7875.

683 Gaitán, J.J., Bran, D.E., Oliva, G.E., Aguiar, M.R., Buono, G.G., Ferrante, D., Nakamatsu, V.,
684 Ciari, G., Salomone, J.M., Massara, V., García Martínez, G., Maestre, F.T., 2017. Aridity
685 and overgrazing have convergent effects on ecosystem structure and functioning in
686 Patagonian rangelands. *Land Degradation and Development*, doi: 10.1002/ldr.2694.

687 Gelaw, A.M., Singh, B.R., Lal, R., 2014. Soil organic carbon and total nitrogen stocks under
688 different land uses in a semi-arid watershed in Tigray, Northern Ethiopia. *Agriculture,*
689 *Ecosystems and Environment* 188, 256-263.

690 Golchin, A., Oades, J.M., Skjemstad, J.O., Clarke, P., 1994. Soil structure and carbon cycling.

691 Australian Journal of Soil Research 32, 1043-1068.

692 Gruber, N., Galloway, J.N., 2008. An Earth-system perspective of the global nitrogen cycle.
693 Nature 451, 293-296.

694 Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global
695 Change Biology 8, 345-360.

696 Hai, L., Li, X.G., Li, F.M., Suo, D.R., Guggenberger, G., 2010. Long-term fertilization and
697 manuring effects on physically-separated soil organic matter pools under a wheat-wheat-
698 maize cropping system in an arid region of China. Soil Biology and Biochemistry 42, 253-
699 259.

700 Halliwell, D.J., Barlow, K.M., Nash, D.M., 2001. A review of the effects of wastewater sodium
701 on soil physical properties and their implications for irrigation systems. Australian Journal of
702 Soil Research 39 1259-1267.

703 Haygarth, P., Ritz, K., 2009. The future of soils and land use in the UK: Soil systems for the
704 provision of land-based ecosystem services. Land Use Policy 26, S187-S197.

705 Haynes, R.J., Williams, P.H., 1999. The influence of stock camping behaviors on the soil
706 microbial and biochemical properties of grazed pastoral soils. Biology and Fertility of Soils
707 28, 253-258.

708 Huang, J., Yu, H., Guan, X., Wang, G., Guo, R., 2016. Accelerated dryland expansion under
709 climate change. Nature Climate Change 6, 166-171.

710 Hugelius, G., Strauss, J., Zubrzycki, S., Harden, J.W., Schuur, E.A.G., Ping, C.L., Schirrmeister,
711 L., Grosse, G., Michaelson, G.J., Koven, C.D., O'Donnell, J.A., Elberling, B., Mishra, U.,
712 Camill, P., Yu, Z., Palmtag, J., Kuhry, P., 2014. Estimated stocks of circumpolar permafrost
713 carbon with quantified uncertainty ranges and identified data gaps. Biogeosciences 11, 6573-

6593.

IIASA/FAO, 2012. Global Agro-ecological Zones (GAEZ v3.0). IIASA, Laxenburg, Austria and
FAO, Rome, Italy.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group
I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change
[Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A.,
Xia, Y., Bex, V., Midgley, P.M. (Eds.)]. Cambridge University Press, Cambridge, UK.

Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T., Wall, D.H., 2002. Ecosystem
carbon loss with woody plant invasion of grasslands. *Nature* 418, 623-626.

Jenny, H., 1941. *Factors of Soil Formation*. McGraw-Hill, New York, NY.

Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its
relation to climate and vegetation. *Ecological Applications* 10, 423-436.

Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta
analysis. *Forest Ecology and Management* 140, 227-238.

Kleber, M., Eusterhues, K., Keiluweit, M., Mikutta, C., Mikutta, R., Nico, P.S., 2015. Mineral–
Organic Associations: Formation, Properties, and Relevance in Soil Environments.
Advances in Agronomy 130, 1-140.

Köchy, M., Hiederer, R., Freibauer, A., 2015. Global distribution of soil organic carbon – Part 1:
Masses and frequency distributions of SOC stocks for the tropics, permafrost regions,
wetlands, and the world. *Soil* 1, 351-365.

Kotzé, E., Sandhage-Hofmann, A., Meinel, J.-A., du Preeza, C.C., Amelung, W., 2013.
Rangeland management impacts on the properties of clayey soils along grazing gradients in
the semi-arid grassland biome of South Africa. *Journal of Arid Environments* 97, 220-229.

737 Lado, M., Ben-Hur, M., 2009. Treated domestic sewage irrigation effects on soil hydraulic
738 properties in arid and semiarid zones: a review. *Soil and Tillage research* 106, 152-163.

739 Laganière, J., Angers, D.A., Paré, D., 2010. Carbon accumulation in agricultural soils after
740 afforestation: a meta-analysis. *Global Change Biology* 16, 439-453.

741 Lal, R., 2001a. Carbon sequestration in dryland ecosystems of West Asia and North Africa. *Land*
742 *Degradation and Development* 13, 45-59.

743 Lal, R., 2001b. Potential of desertification control to sequester carbon and mitigate the
744 greenhouse effect. *Climatic Change* 51, 35-72.

745 Lal, R., 2004a. Carbon sequestration in dryland ecosystems. *Environmental Management* 33,
746 528-544.

747 Lal, R. 2004b. Soil carbon sequestration to mitigate climate change. *Geoderma* 123, 1-22.

748 Lal, R., 2004c. Soil carbon sequestration impacts on global climate change and food security.
749 *Science* 304, 1623-1627.

750 Lal, R., 2009. Challenges and opportunities in soil organic matter research. *European Journal of*
751 *Soil Science* 60, 158-169.

752 Lan, Z., Bai, Y., 2012. Testing mechanisms of N-enrichment induced species loss in a semiarid
753 inner Mongolia grassland: Critical thresholds and implications for long-term ecosystem
754 responses. *Philosophical Transactions of the Royal Society B: Biological Sciences* 367,
755 3125-3134.

756 Lehmann, J., 2007. A handful of carbon. *Nature* 447, 143-144.

757 Lehmann, C.E.R., Anderson, T.M., Sankaran, M., Higgins, S.I., Archibald, S., Hoffmann, W.A.,
758 Hanan, N.P., Williams, R.J., Fensham, R.J., Felfili, J., Hutley, L.B., Ratnam, J., San Jose, J.,
759 Montes, R., Franklin, D., Russell-Smith, J., Ryan, C.M., Durigan, G., Hiernaux, P., Haidar,

760 R., Bowman, D.M.J.S., Bond, W.J., 2014. Savanna vegetation-fire-climate relationships
761 differ among continents. *Science* 343, 548-552.

762 Lehmann, J., Joseph, S., 2015. *Biochar for Environmental Management: Science and*
763 *Technology*, 2nd Edition. Earthscan, London, UK.

764 Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. *Nature* 528, 60-68.

765 Li, D., Niu, S., Luo, Y., 2012. Global patterns of the dynamics of soil carbon and nitrogen stocks
766 following afforestation: a meta-analysis. *New Phytologist* 195, 172-181.

767 Lloyd, J., 1999. The CO₂ dependence of photosynthesis, plant growth responses to elevated CO₂
768 concentrations and their interactions with soil nutrient status. II. Temperature and boreal
769 forest productivity and the combined effects of increasing CO₂ concentrations and increased
770 nitrogen deposition. *Functional Ecology* 13, 439-459.

771 Maestre, F.T., Cortina, J., 2004. Are *Pinus halepensis* plantations useful as a restoration tool in
772 semiarid Mediterranean areas? *Forest Ecology and Management* 198, 303-317.

773 Maestre, F.T., Bowker, M.A., Puche, M.D., Hinojosa, M.B., Martínez, I., García-Palacios, P.,
774 Castillo, A.P., Soliveres, S., Luzuriaga, A.L., Sánchez, A.M., Carreira, J.A., Gallardo, A.,
775 Escudero, A., 2009. Shrub encroachment can reverse desertification in Mediterranean
776 semiarid grasslands. *Ecology Letters* 12, 930-941.

777 Maestre, F.T., Escolar, C., Ladrón de Guevara, M., Quero, J.L., Lázaro, R., Delgado-Baquerizo,
778 M., Ochoa, V., Berdugo, M., Gozalo, B., Gallardo, A., 2013. Changes in biocrust cover drive
779 carbon cycle responses to climate change in drylands. *Global Change Biology* 19, 3835-
780 3847.

781 Maestre, F.T., Delgado-Baquerizo, M., Jeffries, T.C., Ochoa, V., Gozalo, B., Quero, J.L., García-
782 Gómez, M., Gallardo, A., Ulrich, W., Bowker, M.A., Arredondo, T., Barraza-Zepeda, C.,

783 Bran, D., Florentino, A., Gaitán, J., Gutiérrez, J.R., Huber-Sannwald, E., Jankju, M., Mau,
 784 R.L., Miriti, M., Naseri, K., Ospina, A., Stavi, I., Wang, D., Woods, N.N., Yuan, X., Zaady,
 785 E., Singh, B.K., 2015. Increasing aridity reduces soil microbial diversity and abundance in
 786 global drylands. *Proceedings of the National Academy of Sciences of the United States of*
 787 *America* 112, 15684-15689.

788 Maestre, F.T., Eldridge, D.J., Soliveres, S., Kéfi, S., Delgado-Baquerizo, M., Bowker, M.A.,
 789 Gaitán, J., Berdugo, M., Gallardo, A., Lázaro, R., García-Palacios, P., 2016. Structure and
 790 functioning of dryland ecosystems in a changing world. *Annual Review of Ecology,*
 791 *Evolution and Systematics* 47, 215-237.

792 Martínez-Mena, M., Álvarez Rogel, J., Castillo V., Albaladejo, J., 2002. Organic carbon and
 793 nitrogen influenced by vegetation removal in a semiarid Mediterranean soil.
 794 *Biogeochemistry* 61, 309-321.

795 Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural intensification and
 796 ecosystem properties. *Science* 277, 504-509.

797 McLeod, M.K., Schwenke, G.D., Cowie, A.L., Harden, S., 2013. Soil carbon is only higher in
 798 the surface soil under minimum tillage in Vertosols and Chromosols of New South Wales
 799 North-West Slopes and Plains, Australia. *Soil Research* 51, 680-694.

800 Middleton, N., Stringer, L., Goudie, A., Thomas, D., 2011. The Forgotten Billion. MDG
 801 Achievement in the Drylands. UNDP, UNCCD, New York, NY.

802 Middleton, N.J., Sternberg, T., 2013. Climate hazards in drylands: A review. *Earth-Science*
 803 *Reviews* 126, 48-57.

804 Mitchell, J.P., Shrestha, A., Mathesius, K., Scow, K.M., Southard, R.J., Haney, R.L., Schmidt,
 805 R., Munk, D.S., Horwath, W.R., 2017. Cover cropping and no-tillage improve soil health in

806 an arid irrigated cropping system in California's San Joaquin Valley, USA. *Soil and Tillage*
807 *Research* 165, 325-335.

808 Morgan, R.P.C., 2005. *Soil Erosion and Conservation*, 3rd Ed. Blackwell Science, Malden, MA.

809 Morugan-Coronado, A., Arcenegui, V., García-Orenes, F., Mataix-Solera, J., Mataix-Beneyto, J.,
810 2013. Application of soil quality indices to assess the status of agricultural soils irrigated
811 with treated wastewaters. *Solid Earth* 4, 119-127.

812 Naab, J.B., Mahama, G.Y., Koo, J., Jones, J.W., Boote, K.J., 2015. Nitrogen and phosphorus
813 fertilization with crop residue retention enhances crop productivity, soil organic carbon, and
814 total soil nitrogen concentrations in sandy-loam soils in Ghana. *Nutrient Cycling in*
815 *Agroecosystems* 102, 33-43.

816 Nicholson, S.E., 2011. *Dryland Climatology*. Cambridge University Press, Cambridge, UK.

817 Ochoa-Hueso, R., Allen, E.B., Branquinho, C., Cruz, C., Dias, T., Fenn, M.E., Manrique, E.,
818 Pérez-Corona, M.E., Sheppard, L.J., Stock, W.D., 2011. Nitrogen deposition effects on
819 Mediterranean-type ecosystems: an ecological assessment. *Environmental Pollution* 159,
820 2265-2279.

821 Ochoa-Hueso, R., Delgado-Baquerizo, M., Gallardo, A., Bowker, M.A., Maestre, F.T., 2016.
822 Climatic conditions, soil fertility and atmospheric nitrogen deposition largely determine the
823 structure and functioning of microbial communities in biocrust-dominated Mediterranean
824 drylands. *Plant and Soil* 399, 271-282.

825 Ochoa-Hueso, R., Maestre, F.T., de los Ríos, A., Valea, S., Theobald, M.R., Vivanco, M.G.,
826 Manrique, E., Bowker, M.A., 2013. Nitrogen deposition alters nitrogen cycling and reduces
827 soil carbon content in low-productivity semiarid Mediterranean ecosystems. *Environmental*
828 *Pollution* 179, 185-193.

829 Ogle, S.M., Breidt, F.J., Paustian, K., 2005. Agricultural management impacts on soil organic
830 carbon storage under moist and dry climatic conditions of temperate and tropical regions.
831 *Biogeochemistry* 72, 87-121.

832 Palm, C., Blanco-Canqui, H., DeClerck, F., Gatere, L., Grace, P., 2014. Conservation agriculture
833 and ecosystem services: an overview. *Agriculture, Ecosystems and Environment* 187, 87-
834 105.

835 Paul, E.A., 2016. The nature and dynamics of soil organic matter: plant inputs, microbial
836 transformations, and organic matter stabilization. *Soil Biology and Biochemistry* 98, 109-
837 126.

838 Paul, E.A., Paustian, K.H., Elliott, E.T., Cole, C.V., 1997. *Soil Organic Matter in Temperate*
839 *Agroecosystems- Long-term Experiments in North America*. CRC Press, Boca Raton, FL.

840 Paustian, K., Six, J., Elliott, E.T., Hunt, H.W., 2000. Management options for reducing CO₂
841 emissions from agricultural soils. *Biogeochemistry* 48, 147-163.

842 Paustian, K. 2014. Soil: carbon sequestration in agricultural systems. In: Van Alfen, N. (Ed.),
843 *Encyclopedia of Agriculture and Food Systems*. Academic Press, London, UK, p. 140-152.

844 Piñeiro, G., Paruelo, J.M., Oesterheld, M., Jobbágy, E.G., 2010. Pathways of grazing effects on
845 soil organic carbon and nitrogen. *Rangeland Ecology & Management* 63, 109-119.

846 Plaza, C., Fernández, J.M., Pereira, E.I.P., Polo, A., 2012. A comprehensive method for
847 fractionating soil organic matter into pools not protected and protected from decomposition
848 by physical and chemical mechanisms. *CLEAN-Soil Air Water* 40, 134-139.

849 Plaza, C., Courtier-Murias, D., Fernández, J.M., Polo, A., Simpson, A.J., 2013. Physical,
850 chemical, and biochemical mechanisms of soil organic matter stabilization under
851 conservation tillage systems: A central role for microbes and microbial by-products in C

852 sequestration. *Soil Biology & Biochemistry* 57, 124-134.

853 Plaza, C., Giannetta, B., Fernández, J.M., López-de-Sá, E.G., Polo, A., Gascó, G., Méndez, A.,
854 Zaccone, C., 2016. Response of different soil organic matter pools to biochar and organic
855 fertilizers. *Agriculture, Ecosystems & Environment* 225, 150-159.

856 Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and
857 potential. *Global Change Biology* 6, 317-327.

858 Právālie, R., 2016. Drylands extent and environmental issues. A global approach. *Earth-Science*
859 *Reviews* 161, 259-278.

860 Preger, A.C., Kösters, R., Du Preez, C.C., Brodowski, S., Amelung, W., 2010. Carbon
861 sequestration in secondary pasture soils: a chronosequence study in the South African
862 Highveld. *European Journal of Soil Science* 61, 551-562.

863 Puttock, A., Dungait, J.A.J., Macleod, C.J.A., Bol, R., Brazier, R.E., 2014. Woody plant
864 encroachment into grasslands leads to accelerated erosion of previously stable organic
865 carbon from dryland soils. *Journal of Geophysical Research G: Biogeosciences* 119, 2345-
866 2357.

867 Reynolds, J.F., Stafford Smith, D.M., Lambin, E.F., Turner II, B.L., Mortimore, M., Batterbury,
868 S.P.J., Downing, T.E., Dowlatabadi, H., Fernández, R.J., Herrick, J.E., Huber-Sannwald, E.,
869 Jiang, H., Leemans, R., Lynam, T., Maestre, F.T., Ayarza, M., Walker, B., 2007. Goba
870 l desertification: building a science for dryland development. *Science* 316, 847-851.

871 Safriel, U., Adeel, Z., Niemeijer, D., Puigdefabregas, J., White, R., Lal, R., Winslow, M.,
872 Ziedler, J., Prince, S., Archer, E., King, C., Shapiro, B., Wessels, K., Nielsen, T., Portnov,
873 B., Reshef, I., Thonell, J., Lachman, E., McNab, D., 2005. Dryland systems. In: Hassan, R.,
874 Scholes, R., Ash, N. (Eds.), *The Millennium Ecosystem Assessment Series, Ecosystems and*

875 Human Well-being: Current State and Trends, Volume 1. Island Press, Washington, DC, pp
876 623-662.

877 Salinas-Garcia, J.R., Matocha, J.E., Hons, F.M., 1997. Long-term tillage and nitrogen
878 fertilization effects on soil properties of an Alfisol under dryland corn/cotton production.
879 Soil and Tillage Research 42, 79-93.

880 Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber,
881 M., Kogel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner,
882 S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property.
883 Nature 478, 49-56.

884 Schulze, E.D., Freibauer, A., 2005. Carbon unlocked from soils. Nature 437, 205-206.

885 Schuur, E.A.G., McGuire, A.D., Schädel, C., Grosse, G., Harden, J.W., Hayes, D.J., Hugelius,
886 G., Koven, C.D., Kuhry, P., Lawrence, D.M., Natali, S.M., Olefeldt, D., Romanovsky, V.E.,
887 Schaefer, K., Turetsky, M.R., Treat, C.C., Vonk, J.E., 2015. Climate change and the
888 permafrost carbon feedback. Nature 520, 171-179.

889 Senesi, N., Plaza, C., Brunetti, G., Polo, A., 2007. A comparative survey of recent results on
890 humic-like fractions in organic amendments and effects on native soil humic substances. Soil
891 Biology and Biochemistry 39, 1244-1262.

892 Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter
893 accumulation in cultivated and native grassland soils. Soil Science Society of America
894 Journal 62, 1367-1377.

895 Six, J., Elliott, E.T., Paustian, K., 1999. Aggregate and soil organic matter dynamics under
896 conventional and no-tillage systems. Soil Science Society of America Journal 63, 1350-
897 1358.

898 Six, J., Elliott, E.T., Paustian, K., 2000. Soil macroaggregate turnover and microaggregate
899 formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology and*
900 *Biochemistry* 32, 2099-2103.

901 Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic
902 matter: implications for C-saturation of soils. *Plant and Soil* 241, 155-176.

903 Six, J., Ogle, S.M., Breidt, F.J., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to
904 mitigate global warming with no-tillage management is only realized when practiced in the
905 long term. *Global Change Biology* 10, 155-160.

906 Skujins, J., 1991. *Semiarid Lands and Deserts. Soil Resource and Reclamation.* Marcel Dekker,
907 New York, NY.

908 Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic
909 matter: mechanisms and controls. *Geoderma* 74, 65-105.

910 Stavi, I., Argaman, E., Zaady, E., 2016. Positive impact of moderate stubble grazing on soil
911 quality and organic carbon pool in dryland wheat agro-pastoral systems. *Catena* 146, 94-99.

912 Thomas, A.D., 2012. Impact of grazing intensity on seasonal variations in soil organic carbon
913 and soil CO₂ efflux in two semiarid grasslands in southern Botswana. *Philosophical*
914 *Transactions of the Royal Society B* 367, 3076-3086.

915 Thornton, P.K., 2010. Livestock production: recent trends, future prospects. *Philosophical*
916 *Transactions of the Royal Society B* 365, 2853-2867.

917 Tilman, D., Balzer, C., Hill, J., Bafort, B.L., 2011. Global food demand and the sustainable
918 intensification of agriculture. *Proceedings of the National Academy of Sciences of the*
919 *United States of America* 108, 20260-20264.

920 Torn, M.S., Swanston, C.W., Castanha, C., Trumbore, S.E., 2009. Storage and turnover of

921 natural organic matter in soil. In: Huang, P.M., Senesi, N. (Eds.), Biophysico-Chemical
922 Processes Involving Natural Nonliving Organic Matter in Environmental Systems. John
923 Wiley & Sons, Hoboken, NJ.

924 Torn, M.S., Trumbore, S.E., Chadwick, O.A., Vitousek, P.M., Hendricks, D.M., 1997. Mineral
925 control of soil organic carbon storage and turnover. *Nature* 389, 170-173.

926 Trumbore, S., 2009. Radiocarbon and soil carbon dynamics. *Annual Review of Earth and*
927 *Planetary Sciences* 37, 47-66.

928 Turnbull, L., Wainwright, J., Brazier, R.E., 2010. Changes in hydrology and erosion over a
929 transition from grassland to shrubland. *Hydrological Processes* 24, 393-414.

930 UN Environmental Management Group, 2011. Global Drylands: A UN System-wide Response.
931 UN.

932 UNCCD, 1994. United Nations Convention to Combat Desertification in those Countries
933 Experiencing Serious Drought and/or Desertification, Particularly in Africa. A/AC.241/27.

934 UN-DESA-PD, 2015. World Population Prospects: The 2015 Revision, Data Booklet.
935 ST/ESA/SER.A/377.

936 UNEP, 1997. World Atlas of Desertification, 2nd ed. Edward Arnold, London, UK.

937 UNEP-WCMC, 2007. A spatial analysis approach to the global delineation of dryland areas of
938 relevance to the CBD Programme of Work on Dry and Subhumid Lands. Dataset based on
939 spatial analysis between WWF terrestrial ecoregions (WWF-US, 2004) and aridity zones
940 (CRU/UEA; UNEPGRID, 1991). Dataset checked and refined to remove many gaps,
941 overlaps and slivers (July 2014).

942 Van Auken, O.W., 2000. Shrub invasions of North American semiarid grasslands. *Annual*
943 *Review of Ecology and Systematics* 31, 197-215.

944 Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human domination of
945 Earth's ecosystems. *Science* 277, 494-499.

946 Von Lützow, M., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner,
947 B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: mechanisms and their
948 relevance under different soil conditions – a review. *European Journal of Soil Science* 57,
949 426-445.

950 Waldrop, M.P., Zak, D.R., Sinsabaugh, R.L., 2004. Microbial community response to nitrogen
951 deposition in northern forest ecosystems. *Soil Biology and Biochemistry* 36, 1443-1451.

952 Warren, S.D., Eldridge, D.J., 2003. Biological soil crust and livestock in arid ecosystems: are
953 they compatible? In: Belnap, J., Lange, O.L. (Eds.), *Biological Soil Crusts: Structure,*
954 *Function, and Management.* Springer, Berlin, Germany, pp. 401-415.

955 West, T.O., Post, W.M., 2002. Soil organic carbon sequestration rates by tillage and crop
956 rotation: a global data analysis. *Soil Science Society America Journal* 66, 1930-1946.

957 Yang, R., Su, Y.-Z., Wang, T., Yang, Q., 2016. Effect of chemical and organic fertilization on
958 soil carbon and nitrogen accumulation in a newly cultivated farmland. *Journal of Integrative*
959 *Agriculture* 15, 658-666.

960 Zak, D.R., Freedman, Z.B., Upchurch, R., Steffens, M., Kögel-Knabner, I., 2016. Anthropogenic
961 N deposition increases soil organic matter accumulation without altering its biochemical
962 composition. *Global Change Biology* 23, 933-944.

963 Zhang, F., Li, C., Wang, Z., Glidden, S., Grogan, D.S., Li, X., Cheng, Y., Frolking, S., 2015.
964 Modeling impacts of management on farmland soil carbon dynamics along a climate
965 gradient in Northwest China during 1981-2000. *Ecological Modelling* 312, 1-10.

FIGURE CAPTIONS

Figure 1. Map of dryland areas in accordance with the United Nations Convention to Combat Desertification (UNCCD) and Convention on Biological Diversity (CBD) definitions (UNEP-WCMC, 2007). Based on the aridity index (AI), or ratio of total annual precipitation to potential evapotranspiration, drylands are divided into hyperarid (AI less than 0.05), arid (AI within the range from 0.05 to 0.2), semiarid (AI from 0.2 to 0.5), and dry subhumid regions (AI from 0.5 to 0.65).

Figure 2. Simplified global C cycle and soil C stocks in drylands (data from Safriel et al., 2005; IPCC, 2013).

Figure 3. Organic matter transformation, stabilization, and protection from decomposition in soils (redrawn from Courtier-Murias et al., 2013, with permission from Elsevier).

Figure 4. Soil organic matter (OM) fractions isolated from a dryland soil and their links to conceptual mechanisms of stabilization and protection from decomposition (reprinted from Plaza et al., 2013, with permission from Elsevier).

Figure 5. Typical patchy vegetation of drylands (image by Fernando T. Maestre).

Figure 6. Biological soil crust (image by Fernando T. Maestre).

Figure 7. Total organic C content of semiarid soils under no tillage, chisel plow, and moldboard plow for 23 years (1986-2009) either unamended or organically amended for 2 years (2007-2009). Within the same tillage system, different lowercase letters indicate significant differences according to the LSD test at the 0.05 level. Within the same amendment system, different uppercase letters indicate significant differences according to the LSD test at the 0.05 level (data from Haller, 2011).

Figure 8. Main strategies to restore, preserve, and enhance soil organic matter in drylands (Lal, 2004a, 2009).

Figure 9. Grain, straw, and total dry matter yield of winter barley (*Hordeum vulgare* L.) grown on semiarid soils either unamended (C) or amended with mineral fertilizer (MF), 20 and 80 Mg ha⁻¹ of sewage sludge compost (CS20 and CS80), and 20 and 80 Mg ha⁻¹ of thermally dried sewage sludge (TS20 and TS80). Within the same yield component, different letters indicate significant differences based on LSD test at the 0.05 level (data from Fernández et al., 2009).

Figure 10. Carbon biomass cycle and sequestration in biochar. Plants store atmospheric C by photosynthesis in their biomass. Biomass C is released back into the atmosphere by plant respiration or incorporated into the soil. Soil organic C decompose and release C into the atmosphere. The pyrolysis of biomass generates energy and biochar, with very stable C structures that decompose very slowly and can be stored in soils for much longer than the biomass C from which it originates (Lehmann, 2007; Lehmann and Joseph, 2015).

Figure 11. Free, intra-macroaggregate, intra-microaggregate, and mineral-associated organic C content of soils either unamended (UN), amended with municipal solid waste compost (MC), or amended with sewage sludge (SS), without or with biochar (BC). Error bars indicate pooled

standard error. Different letters indicate significant differences according to LSD test at the 0.05 level (reprinted from Plaza et al., 2016, with permission from Elsevier).

Figure 12. Organic C content (on whole soil basis) of no-tillage and chisel tilled semiarid soils and in the corresponding free (FR), intra-macroaggregate (MA), intra-microaggregate (MI), mineral-associated (Min), and dissolved (DOM) organic matter fractions. Mean $\pm$ standard error, $n = 3$ (data from Plaza et al., 2013).

Figure 13. Diffusion edited ^1H HR-MAS NMR spectra of microbes cultured from soil, barley straw, semiarid soils under chisel tillage for 25 years, and their respective free, intramacroaggregate, intra-microaggregate, and mineral-associated organic matter (OM) pools. General assignments of chemical shift ranges are as follows: (a) CH_3 and CH_2 (0.6-1.3 ppm); (b) CH_3 and CH_2 near O and N (1.3-2.9 ppm); (c) O-alkyl, mainly from carbohydrates and lignin (2.9-4.1 ppm); (d) α -H from peptides (4.1-4.8 ppm); (e) aromatic, from lignin and peptides (6.2-7.8 ppm); and (f) amide in peptides (7.8-8.4 ppm). It is noteworthy that the spectrum of the mineral-associated OM pool is very similar to that of soil microbial cultures, whereas the spectra of the other OM fractions more closely resemble that of barley straw (spectra redrawn from Plaza et al., 2013 and Courtier-Murias et al., 2013 with permission from Elsevier).

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Whole soil
(8-mm-sieved, air-dried)

Dissolved OM
(unprotected)

Free OM
(unprotected)

Intra-macroaggregate OM
(weakly protected by
physical mechanisms)

Intra-microaggregate OM
(strongly protected by
physical mechanisms)

Mineral-associated OM
(protected by chemical
mechanisms)

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