A historical perspective on soil organic carbon in Mediterranean cropland (Spain, 1900–2008)

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Eduardo Aguilera
Universidad Politécnica de Madrid

Gloria I. Guzmán
Universidad Pablo de Olavide

Jorge Álvaro-Fuentes
Spanish National Research Council

Juan Infante-Amate
Universidad Pablo de Olavide

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A historical perspective on soil organic carbon in Mediterranean cropland (Spain, 1900–2008)

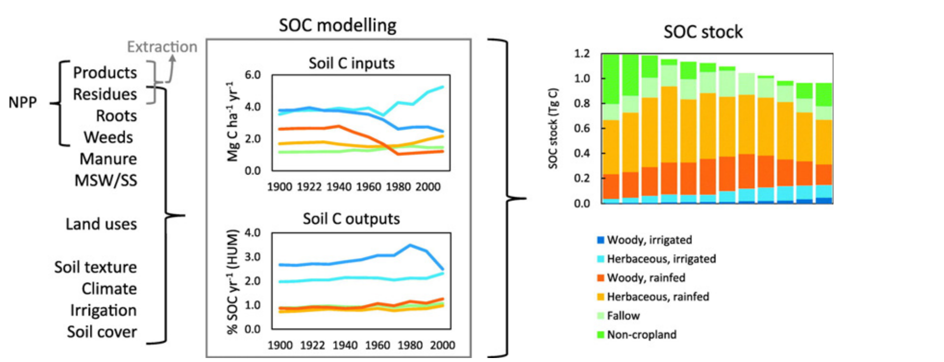
Eduardo Aguilera [a,*], Gloria I. Guzmán [a], Jorge Álvaro-Fuentes [b], Juan Infante-Amate [a], Roberto García-Ruiz [c], Guiomar Carranza-Gallego [a], David Soto [a], Manuel González de Molina [a]

[a] *Agro-ecosystems History Laboratory, Universidad Pablo de Olavide, 41013 Sevilla, Spain*
[b] *Departamento de Suelo y Agua, Estación Experimental de Aula Dei, Consejo Superior de Investigaciones Científicas (EEAD-CSIC), 50059 Zaragoza, Spain*
[c] *Departamento de Biología Animal, Biología Vegetal y Ecología, Universidad de Jaén, 23071 Jaén, Spain*

HIGHLIGHTS

- Long-term land use, management and climate impacts on cropland SOC were modeled.
- Historical data and literature coefficients used to estimate NPP and soil C inputs
- Continuous cropland SOC declines along the studied period.
- Major drivers of SOC loss shifted from land use, to management to climatic changes.
- High impact of weed declines on SOC in woody crops; mixed effects of irrigation.

GRAPHICAL ABSTRACT



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ABSTRACT

Soil organic carbon (SOC) management is key for soil fertility and for mitigation and adaptation to climate change, particularly in desertification-prone areas such as Mediterranean croplands. Industrialization and global change processes affect SOC dynamics in multiple, often opposing, ways. Here we present a detailed SOC balance in Spanish cropland from 1900 to 2008, as a model of a Mediterranean, industrialized agriculture. Net Primary Productivity (NPP) and soil C inputs were estimated based on yield and management data. Changes in SOC stocks were modeled using HSOC, a simple model with one inert and two active C pools, which combines RothC model parameters with humification coefficients. Crop yields increased by 227% during the studied period, but total C exported from the agroecosystem only increased by 73%, total NPP by 30%, and soil C inputs by 20%. There was a continued decline in SOC during the 20th century, and cropland SOC levels in 2008 were 17% below their 1933 peak. SOC trends were driven by historical changes in land uses, management practices and climate. Cropland expansion was the main driver of SOC loss until mid-20th century, followed by the decline in soil C inputs during the fast agricultural industrialization starting in the 1950s, which reduced harvest indices and weed biomass production, particularly in woody cropping systems. C inputs started recovering in the 1980s, mainly through increasing crop residue return. The upward trend in SOC mineralization rates was an increasingly important driver of SOC losses, triggered by irrigation expansion, soil cover loss and climate change-driven temperature rise.

* Corresponding author at: Agro-ecosystems History Laboratory, Universidad Pablo de Olavide, Ctra. de Utrera, km. 1, 41013 Sevilla, Spain.
E-mail address: emagufer@upo.es (E. Aguilera).

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1. Introduction

Carbon (C) dynamics in cropland soils is a hot topic in Mediterranean agriculture in the context of climate change. On the one hand, soil organic carbon (SOC) is highly sensitive to management practices in these environments (Aguilera et al., 2013), and soil C sequestration can offset a large share of the full life cycle greenhouse gas (GHG) emissions of crop production (e.g. Bosco et al., 2013; Aguilera et al., 2015a, 2015b; Guardia et al., 2016). On the other hand, Mediterranean cropping systems are characterized by low SOC levels, resulting particularly vulnerable to climate change (Zalidis et al., 2002; Iglesias et al., 2011). This last condition stresses the need for increasing SOC levels to improve soil quality and fertility (Diacono and Montemurro, 2010).

Global cropland area expanded about 4-fold since 1700, up to ca. 20% of the vegetated area (Pongratz et al., 2008). In parallel, agroecosystems intensified to meet increases in population density (Ellis et al., 2013), culminating in a radical socio-metabolic transformation, with the transition from traditional organic, solar-based systems to industrial, fossil fuel based systems (Krausmann et al., 2008). The former usually operated at the local scale, relying on solar fluxes and internal biomass recycling as sources of energy and fertility. The latter are intensified through imports of fossil fuel-based industrial inputs and international or interregional trade to optimize the conditions for commodity production in a context of a global market economy (Guzmán and González de Molina, 2017; Gingrich et al., 2017). These structural characteristics shape the C cycle through effects on the type and quantity of soil C inputs and on the biotic and abiotic factors controlling C losses. Soil C inputs can potentially increase with industrialization due to a higher overall biomass production and a lower use of crop residues for animal feeding (e.g. Wiesmeier et al., 2014). Modern crops, however, usually have higher harvest indices, which reduces the production of residue relative to the main product (Johnson et al., 2006). In addition, weed biomass is more effectively suppressed in modern cropping systems (Guzmán et al., 2014), and root growth in relation to aerial biomass is usually reduced (e.g. Chirinda et al., 2012). SOC mineralization is also affected by the changes in management practices such as tillage, irrigation and fertilization (Sainju et al., 2013; Shang et al., 2015).

Along with historical management changes, SOC dynamics are affected by shifts in environmental conditions associated to global change, particularly temperature increase, which would boost litter decay (Gregorich et al., 2017) and SOC mineralization (Davidson and Janssens, 2006), potentially representing a positive feedback to climate change. On the other hand, possible reductions in precipitation in Mediterranean areas (Giorgi and Lionello, 2008) could increase water limitation of SOC mineralization.

SOC dynamics in modern conventional systems have been often compared to those of modern organic and/or low-input systems (Gattinger et al., 2012; Aguilera et al., 2013). However, specific studies on traditional organic cropping systems are very scarce, particularly at large spatial scales. Most large-scale assessments of SOC dynamics are based on crop-soil process-based simulations, validated with soil and yield data from databases such as EUROSTAT or FAOSTAT (e.g. Ciais et al., 2010; Bondeau et al., 2007). Most of these databases do not provide data before mid-20th century, or specific information on many management practices, a problem that can be overcome in studies covering smaller areas with better statistical information (e.g. Parton et al., 2015).

Spanish agriculture experienced vast technological and structural changes along the 20th century. During the second half of the century, there was a large increase in land and animal productivity, which was used to feed an increasing population, to increase the share of animal products in the diet, and to raise exports of high-value crop products (Lassaletta et al., 2014; Soto et al., 2016). Recent assessments have shown some of the biophysical costs of these productivity gains. The reliance on external and total energy consumed led to a significant decrease in the energy return on investment (EROI) (Guzmán et al., 2017), a growing dependence on feed biomass imports (Soto et al., 2016), a large nitrogen (N) surplus (Lassaletta et al., 2014) and a strong pressure on scarce water resources (Duarte et al., 2014). The main aims of this study were to analyze cropland SOC dynamics in Spain, used here as a model Mediterranean country, in the long-term (+100 years), and to identify the main drivers responsible for the observed trends. The specific objectives were: (i) to build and test a simplified SOC model for its use in historical studies; (ii) to reconstruct NPP and soil C inputs from 1900 to 2008; (iii) to simulate SOC stock changes from 1900 to 2008; and (iv) to test the sensitivity of the model outputs to changes in key model parameters.

2. Methods

2.1. Study site characteristics

Climate in Spain is mostly Mediterranean, with hot, dry summers and wet, mild autumns and winters. Severe water deficit during the summer (Fig. 1a) is one of the features controlling crops distribution and management. There is a strip of temperate climate in the northern coast, and a gradient of dryness towards the South-East (Fig. 1d).

Annual mean precipitation during the 20th century in Spain ranged from 500 to 900 mm, with no clear trend (Fig. 1b). Mean temperature increased from 12.3 °C in the 1900–1909 period to 13.8 °C in the 2000–2002 period (Fig. 1c, e, f), corresponding to an average increase rate of 0.17 °C per decade, which can be compared to the 0.1 °C decadal average global land warming estimated for the 1901–2012 period (Hartman et al., 2013). These changes have resulted in increasing drought severity during the last 50 years (Vicente-Serrano et al., 2014).

The main soil orders in Spain are Entisols and Inceptisols, which account for more than three-quarters of the total national surface area (Gómez-Miguel and Badía-Villas, 2016). National means and standard deviations (in parenthesis) for pH, soil organic matter, and sand, silt and clay proportions (%) are 7.47 (1.49), 2.53 (2.87), 51.77 (19.99), 26.50 (14.73) and 21.77 (10.98), respectively (López Arias and Grau Corbí, 2005).

Area and production values for each crop type-management category were retrieved from the Agricultural Statistics Yearbooks, available online at MAPAMA (2017). In some cases, area and production values in rainfed and irrigated land had to be adjusted to match the total values provided in the source and the total irrigated area. Outliers in the data were also disregarded. The estimation of the total irrigated area and the segregation of the irrigated area by irrigation types was based on various official reports (MAICOP, 1904; MF, 1918; MAGRAMA, 2015) and secondary sources (Calatayud and Martínez-Carrión, 2005). Cropland area increased from 33% in 1900 to 41% in 1970, decreasing down to 34% by 2008 (Fig. 2a). Herbaceous crops represent the majority of cropland area (Fig. 2a), but the share of woody crops is also very significant (ranging from 18% in 1900 to 28% in 2008). Fallow land was highest in 1960 (36% of herbaceous crops area), and lowest in 2000 (24%). Irrigated area increased from 6% to 19% of cropland from 1900 to 2008, with a growing share of sprinkler irrigation systems since 1970 and of drip irrigation systems since 1990 (Fig. 2b, c).

2.2. Soil organic carbon model description

Humified Soil Organic Carbon (HSOC) model is an adaptation of RothC model (Coleman and Jenkinson, 1996), consisting in its simplification into two active SOC pools; fresh organic matter, (FOM) and humus (HUM), and one inactive pool (IOM) (Fig. 3). In HSOC model, the three labile C pools in RothC (resistant plant material, decomposable plant material, and microbial biomass) are merged into a single pool (FOM). This allows for the reduction of internal feedbacks and thus for an easier interpretation of the model functioning. The simplification of the model also allows for a better integration of factors that have an

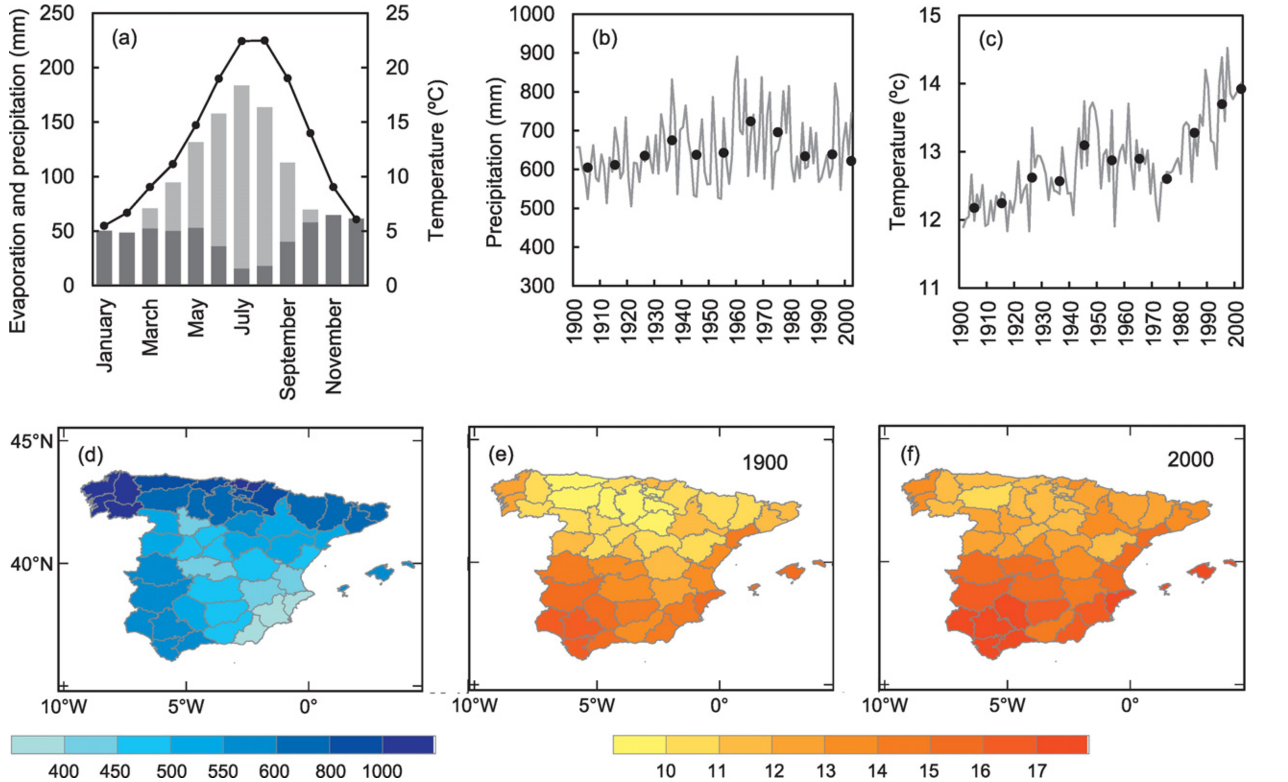

Fig. 1. Agro-climatic features of Spain during the studied period. (a) 1901–2002 monthly average of potential evaporation (light bars), precipitation (dark bars) and temperature (dots and lines), (b) annual (lines) and decadal (dots) average precipitation (mm), (c) annual (lines) and decadal (dots) average air temperature, (d) provincial distribution of annual average precipitation in 1900–2002 (mm), (e) provincial distribution of annual average air temperature in 1900–1909 (°C), and (f) provincial distribution of annual average air temperature in 2000–2002.
Data from Goerlich Gisbert (2012).

influence on SOC dynamics but are not considered in RothC, particularly the effect of C input quality on humification rates.

Both active pools in HSOC model follow first-order kinetics, *FOM* with a fast turnover rate and *HUM* a slow one. Decomposition rates of both pools are controlled by agro-climatic factors. The inputs to *HUM* are calculated from annual soil C inputs using input-specific humification coefficients (H_i). Inputs to *FOM* are calculated as the total C inputs applied to the soil minus those that are humified.

Starting from year *0*, SOC content C in year *t* is given by:

$$SOC_t = IOM + HUM_t + FOM_t SOC_t = IOM + HUM_t + FOM_t \quad (1)$$

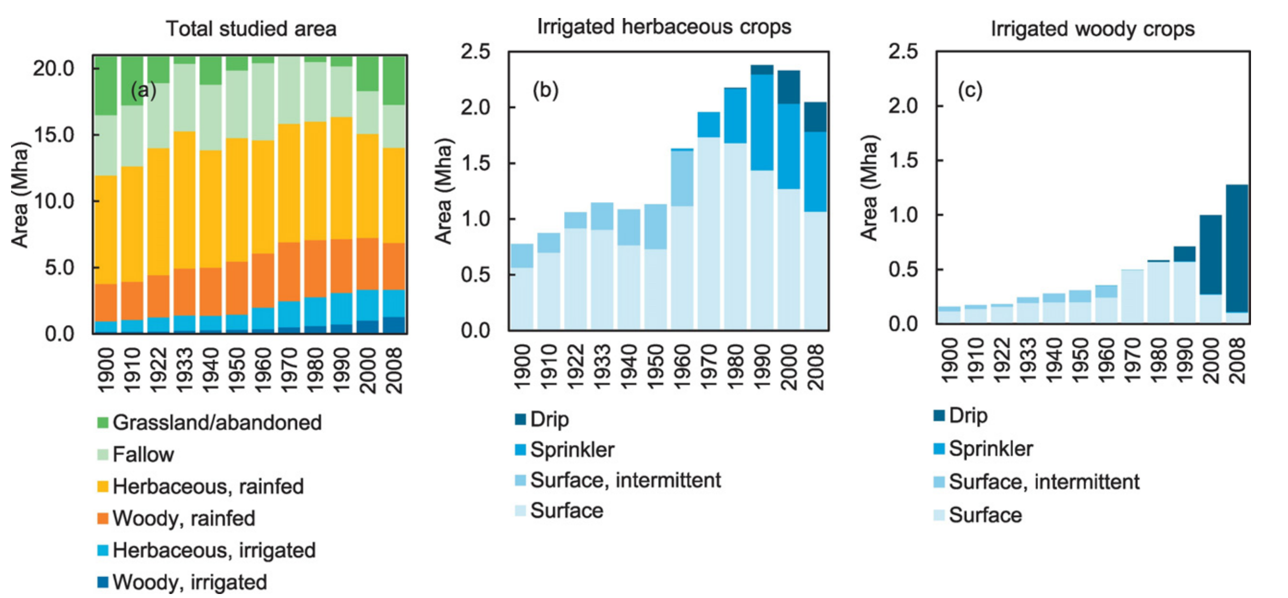

Fig. 2. Land use distribution in the studied area of Spain, corresponding to the maximum cropland area over the study period. (a) Area distribution by land use type and presence of irrigation, (b) irrigation types in herbaceous crops, and (c) irrigation types in woody crops.
Sources: MAPAMA (2017) for land uses, and MAICOP (1904), MF (1918), Calatayud and Martínez-Carrión (2005) and MAGRAMA (2015) for irrigation types.

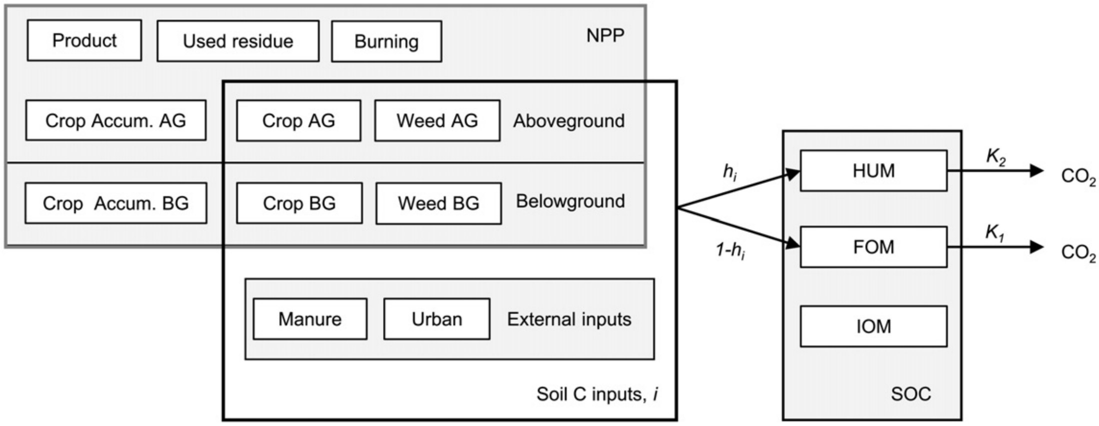

Fig. 3. Basic structure of HSOC model, and components of net primary production (NPP) and soil carbon (C) inputs. NPP components are categorized according to their source (crop or weed), their placement (aboveground, *AG*, and belowground, *BG*), and their final use. The biomass of aboveground crop residues is divided into *Crop AG* (unharvested and applied to the soil), *burning* (burnt on field), *used residues* (extracted from the field) and *accumulated* (incorporated into woody tissues and remaining in the field). $h_{i:}$ humification coefficient of each C input I, *HUM*: slow turnover soil organic carbon (SOC) pool; *FOM*: fast turnover SOC pool; *IOM*: inert SOC pool; K_1 and K_2 stand for annual decomposition rates.

$$HUM_t = HUM_0 \cdot e^{-K_2 \cdot t} + \sum_{i=0} \frac{I_i \cdot h_i}{K_2} \cdot \left(1 - e^{-K_2 \cdot t}\right) \tag{2}$$

$$FOM_t = FOM_0 \cdot e^{-K_1 \cdot t} + \sum_{i=0} \frac{I_i \cdot (1 - h_i)}{K_1} \cdot \left(1 - e^{-K_1 \cdot t}\right) \tag{3}$$

where K_1 and K_2 are the decomposition rates of *FOM* and *HUM* during the studied period, respectively, *I* is the amount of each C input *i* applied each year, and h_i is its corresponding humification coefficient. IOM is calculated following Falloon et al. (1998) equation, which was applied to the average SOC levels for Spanish cropland in 2008 (Rodríguez-Martín et al., 2016).

Total SOC stock at equilibrium is given by:

$$SOC_{eq} = IOM + \frac{\sum_{i=0} I_i \cdot h_i}{K_2} + \frac{\sum_{i=0} I_i \cdot (1 - h_i)}{K_1} \tag{4}$$

Tillage is an important factor controlling SOC dynamics, but it was not included in the model due to the following reasons: 1) Tillage is not included in RothC model, which is the basis for HSOC model and has been validated without this factor. 2) Changes in tillage practices produce a vertical redistribution of SOC (e.g. Manley et al., 2005; Luo et al., 2010). This increases the uncertainty of the estimates of total SOC changes, that are strongly dependent on sampling depth; 3) Primary data regarding the historical changes in tillage practices in Spain is lacking, as far as we know.

2.3. Net primary production

Net Primary Production (NPP) in Spanish cropland was estimated as detailed in Soto et al. (2016) and Guzmán et al. (2017). In short, area and yield data of all crops cultivated in Spain from 1900 to 2008, collected from official statistics (MAPAMA, 2017), and expressed as 5-year averages in 12 selected time points. In this study, it was also distinguished between rainfed and irrigated crops. Total aboveground (AG) crop biomass production was estimated by applying dry matter coefficients and harvest indices (Guzmán et al., 2014). Harvest indices of traditional and modern cereal varieties were distinguished.

Allometric functions ("Fixed root:shoot ratios") are the most common approach for the estimation of root biomass production in GHG inventories and large-scale NPP or C assessments (Johnson et al., 2006; Taghizadeh-Toosi et al., 2016; Niedertscheider et al., 2016), and they have also been used in biomass and energy analyses of Spanish agriculture (Soto et al., 2016; Guzmán et al., 2017). There is increasing evidence, however, that root:shoot ratios are decreased by yield-promoting management changes, particularly by inputs of synthetic nutrients (Andrews et al., 2001; Chirinda et al., 2012; Poeplau, 2016; Poeplau et al., 2016; Grechi et al., 2007) and irrigation (Buwalda, 1993; Kozlowski and Pallardy, 2002; Gan et al., 2009; Wilcox et al., 2016). Consequently, simulation studies suggest that using fixed root:shoot ratios overestimates the response of roots to shoot productivity changes (Poeplau, 2016), and that using a fixed amount of root mass ("Fixed root mass") may lead to better fit with experimental data (Taghizadeh-Toosi et al., 2016). Other studies, however, suggest that root-shoot ratios of crops may have even increased along recent history (Wiesmeier et al., 2014). Given this controversy, root biomass production was estimated as the average of the "fixed root:shoot ratio" and the "fixed root mass" approaches. The reference root mass value for the fixed root biomass approach was calculated by applying reference root:shoot ratios (Table A.1) to the aboveground production values of year 2000. In the case of traditional wheat varieties, the root:shoot ratio was set at 38% above that of modern varieties, based on our own data from a field experiment (Carranza et al., 2016), and in line with other published studies (Siddique et al., 1990; Bektas et al., 2016). The effects of using fixed root:shoot ratios and fixed root mass approaches were assessed in a sensitivity analysis.

Above-ground and below-ground weed biomass production were taken into account in the estimation of cropland NPP (Soto et al., 2016), using data on modern organic agriculture for the 1900–1950 period, modern conventional agriculture for the 1980–2008 period (Guzmán et al., 2014, 2017), and linearly interpolating in the intermediate time points. A scenario without weeds was incorporated in the sensitivity analysis.

Carbon contents of crop products were taken from Wolf et al. (2015). Modeling studies usually assume a general value for the C content of inputs (e.g. Wolf et al., 2015; Nair, 2012). In this study, specific values were used for crop species or categories, depending on data availability (Table A.2).

2.4. Soil C inputs

The annual plant C input to the soil was estimated by subtracting from total NPP the biomass extracted, burned and accumulated. Extracted biomass includes all crop production and the fraction of crop residues that is used, including harvested and grazed residues. Harvested crop and residue data were directly taken from statistics, while grazed biomass was estimated with a feed balance approach (Soto et al., 2016).

The fraction of residues burned in the field was also taken from Soto et al. (2016), with modifications for cereals and industrial crops (Table A.3). Accumulated biomass refers to the fraction of the biomass of woody crops that is annually incorporated into permanent woody tissues.

External soil C inputs include manure and urban waste C inputs. Manure includes excretion of grazing animals and applications of managed manure. The amount of manure produced, in terms of N, was estimated though a mass-balance approach, by subtracting gross N production in slaughtered animals and livestock products (calculated using coefficients from Bodirsky et al., 2012) from total N ingestion by livestock, estimated with biomass intake values estimated in Soto et al. (2016) and N contents from García-Ruiz et al. (in prep). The distribution of excreted N among animal species and fates is described in Appendix B. N losses associated to manure management were estimated at 35.7% of excreted N (Pardo et al., 2015). Manure N was converted to C using C:N ratios from the literature (Table 1).

The application of urban waste to cropland is reported by official statistics since 1990, expressed as N (MAPAMA, 2017 BNAE), which was converted to C using the data form Table 1. For the previous period, it was assumed that the per capita agricultural use of urban waste was the same as in 1990.

2.5. Humification coefficients

Humification coefficients (H_i) indicate the fraction of soil C inputs entering the slow turnover *HUM* pool. H_i are specific for each C input type *i*, and they are calculated from basal humification coefficients, h_i, modified by a soil texture modifying factor, *d*, as follows:

$$H_i = h_i \cdot d \tag{5}$$

The literature was searched for input-specific humification coefficients (h_i) derived from long-term (>4 years) experiments (Table 2). The definition and the calculation methodology varies among studies, which contributes to the variability found in the data. In many studies, a "carbon retention" coefficient is estimated by calculating the relative amount of cumulative C input retained in the soil after a certain period, either through ^{13}C isotope analyses (e.g. Bolinder et al., 1999) or from the slope of the SOC increase versus the cumulative C input (e.g. Maillard and Angers, 2014). This approach is sensitive to SOC decomposition rate and study duration. In another group of studies, the C input and SOC dynamics data were fitted into first-order dynamics SOC models, usually with one active SOC compartment (e.g. Bayer et al., 2006), but also with two compartments (e.g. Andren and Kätterer, 1997). In spite of these conceptual differences, we did not find significant differences among the *h* values estimated with the different approaches (data not shown), and the average study duration was similar among the studied categories (Table 2).

Lower *h* values for herbaceous crop residues than for roots (Table 2) are in line with many studies comparing both types of materials (e.g. Bolinder et al., 1999; Rasse et al., 2005; Kätterer et al., 2011; Berti et al., 2016). In the case of pruning residues of woody crops, the relatively high value is in line with studies in forest soils (Polglase et al., 2000; Zhou et al., 2016). Values for external amendments are in agreement with the median C retention of 31% (N = 25) observed by Aguilera et al. (2013) for a wide range of external C inputs applied to soils under Mediterranean conditions. In spite of this, Maillard and Angers (2014) found a global average C retention coefficient for manure of only 12%, which was used to build a scenario in the sensitivity analysis.

Soil texture is usually assumed to influence the stabilization of SOC (e.g. Poeplau et al., 2015), although it may influence SOC dynamics through SOC mineralization (Saffih-Hdadi and Mary, 2008). In HSOC model, humification coefficients are modulated by the texture modifying factor *d*, which is adapted from Coleman and Jenkinson (1996) equation for the partitioning of plant material between CO_2 and HUM and BIO pools in RothC.

$$d = 3.51/\left(1.67 \cdot \left(1.85 + 1.6 \cdot e^{-0.0786 \cdot \mathrm{CF}}\right)\right) \tag{6}$$

where CF is the clay fraction of the soil (%). As in RothC, soil texture in HSOC model is only included as a modifying factor of humification, not on SOC mineralization. However, it is worth noting that texture indirectly affects the mineralization rates through its effect on soil moisture.

2.6. Decomposition rates of SOC pools

Decomposition rates K_1 and K_2 are based on RothC model approach and parameters. Decomposition rate constants k_1 and k_2 are modulated each month *m* by modifying factors, as follows:

$$K = \sum_{m=12} (k \cdot a_m \cdot b_m \cdot c_m \cdot t) \tag{7}$$

where *a* is the rate modifying factor for temperature, *b* for moisture, *c* for soil cover, and t is the time step, corresponding to 1/12. k_1 was set at 48%, so that FOM matched the SOC fraction represented at equilibrium by the equivalent pools in RothC. k_2 was set at 0.02, as in RothC. The equations for *a*, *b*, and *c* modifying factors were taken from RothC (Coleman and Jenkinson, 2014).

Monthly average climate data for the 1901–2002 period in 46 out of 50 Spanish provinces (excluding Canary Islands, Ceuta and Melilla) was gathered from Goerlich Gisbert (2012), who transformed the grid data from the CRU TS 2.1 database to the provincial level. Potential evapotranspiration was calculated with the Hargreaves equation (Hargreaves and Samani, 1985). A sensitivity analysis scenario was also built using the average climate data (1901–2002) for the whole simulation period, in order to isolate the effect of changes in climate on SOC changes.

The crop area and production data was not spatially disaggregated for the whole study period, so the climate modifying factors had to be aggregated on a national level in order to run the model simulations. This aggregation was calculated as a weighted average of the provincial climate modifying factors. Weighting was based on the relative provincial distribution of crop-management categories in year 2000 (MAPAMA, 2017, Table A.4). Crop-management categories segregate

Table 1
Mean, standard deviation (SD), number of studies (N) of C:N ratios of external C inputs. See text for definitions of categories.

	Mean	SD	N	Sources
Manures "ready to apply"				
Farmyard manure	21.2	9.3	34	[1–11]
Pig manure	13.2	3.3	7	[3, 4, 8, 14]
Poultry litter	8.6	2.6	19	[3, 4, 8, 12–14]
Pig slurry	4.2	3.5	77	[15]
Bovine slurry	7.1	3.3	8	[3, 9, 16, 17]
Manures "as excreted"				
Bovine	19.1	4.4	18	[1, 10, 17–20]
Ovine/caprine	12.4	5.6	7	[19, 21, 22]
Pig	9.1	3.7	8	[8, 23–27]
Poultry	6.9	1.1	3	[8, 13, 26, 12]
Urban wastes				
Sewage sludge	7.3	1.8	20	[4, 6, 28, 29]
Municipal solid waste, composted	30.6	25.5	8	[4, 28, 29]

[1] Castellanos-Navarrete et al. (2015), [2] Chastain and Moore (2014), [3] Chadwick et al. (2000), [4] ECN (2017), [5] Ghosh et al. (2012), [6] Iakimenko et al. (1996), [7] Miller et al. (2003), [8] Mishima et al. (2012), [9] Pettygrove et al. (2009), [10] Tittonell et al. (2010), [11] Yamulki (2006), [12] Edwards and Daniel (1992), [13] Griffiths (2011), [14] Xu et al. (2017), [15] Antezana et al. (2016), [16] Amon et al. (2006), [17] Triberti et al. (2008), [18] Chen et al. (2003), [19] Jarvis et al. (1995), [20] Thomsen et al. (2013), [21] Ma et al. (2007), [22] Mafongoya et al. (2000), [23] Jacobs et al. (2011), [24] Jarret et al. (2012), [25] Jorgensen et al. (2013), [26] Kirchmann and Witter (1992), [27] Vu et al. (2009), [28] Mondini et al. (2017), [29] Plaza et al. (2016).

Table 2
Mean, standard deviation (SD), number of studies (N), of the humification coefficients (*h*) for C input categories (% of applied C). The mean duration of the experiments is also shown.

	Mean (%)	SD (%)	N	Mean duration (years)	References
Herbaceous residues	11.5	4.3	15	24	[1–9]
Herbaceous residue + roots	16.1	7.8	11	29	[10–20]
Pruning residues	32.5	3.5	2	Not available	[21,22]
Roots	21.8	7.7	13	30	[2–4, 6, 7, 23–26]
Extra-root C[a]	8.0			Not available	Calculated from [26]
Manure	25.4	10.5	11	34	[1, 3–6, 9, 24, 27, 28]
Sewage sludge	38.6	13.7	5	35	[1, 4, 6, 27]

References: [1] Andren and Kätterer (1997), [2] Barber (1979), [3] Berti et al. (2016), [4] Boiffin et al. (1986), [5] Delas and Molot (1983), [6] Kätterer et al. (2011), [7] Plenet et al. (1993), [8] Thomsen and Christensen (2010), [9] Thomsen and Christensen (2004), [10] Bayer et al. (2006), [11] Bolinder et al. (1999) (from Balesdent et al., 1990), [12] Buyanovsky and Wagner (1998), [13] Campbell et al. (1991), [14] Kong et al. (2005), [15] Lovato et al. (2004), [16] Gregorich et al. (1995), [17] Gregorich et al. (1996), [18] Poeplau et al. (2015), [19] Saffih-Hdadi and Mary (2008), [20] Vieira et al. (2009), [21] Sofo et al. (2005), [22] Bosco et al. (2013), [23] Angers et al. (1995), [24] Bertora et al. (2009), [25] Bolinder et al. (1999), [26] Brock et al. (2012), [27] Bhogal et al. (2007), [28] Maillard and Angers (2014).

[a] Own calculation from root turnover and rhizodeposits data.

production and area data based on main crop types (winter cereals, summer cereals, forage, other herbaceous, fruits, treenuts, olive, grapevine fallow and non-cropland) and two management types (rainfed and irrigated).

Annual irrigation water inputs were assumed to be 750, 650 and 500 mm for surface, sprinkler and drip irrigation systems, respectively (Corominas, 2010). Water inputs for intermittent irrigation were estimated as one third of those of surface irrigation systems. Water inputs were distributed monthly taking into account the crop-growing period and the water balance in each month.

Monthly soil cover for the selected crop-management categories was estimated according to our own field experience (Table A.5). The monthly average soil cover in woody crops was modeled by defining two extreme categories representing systems with and without cover crops (Table A.5), which were weighted according to an index (Table A.6) based on the amount of biomass produced by the cover crop, taken from Soto et al. (2016).

2.7. Model application

The model was first tested by comparing it to RothC and experimental data under Mediterranean conditions (Appendix B), including experiments with rainfed barley (Álvaro-Fuentes et al., 2012), rainfed cereal rotations (Sombrero and de Benito, 2010), and rainfed olives (Nieto et al., 2010). A good agreement between modeled and observed SOC values was observed, as well as between HSOC and RothC model outputs. In particular, we verified that the dynamics of the FOM pool were very similar to those of the sum of the three labile pools in RothC (DPM, RPM and BIO), indicating the validity of the unification of these pools into a single pool that was implemented in HSOC. After validation, the model was applied to reconstruct cropland SOC in Spain. The model was applied to the 0–30 cm layer of the soil. *Total studied area* corresponds to the maximum cropland area in Spain during the studied period, which was 20.9 Mha in 1970 (Fig. 2a). As cropland area changes along time, the remaining area (named "non-cropland") was modeled as the weighted average of grassland and forestland in the country, with area and biomass production data from Soto et al. (2016). Woody belowground biomass values reported by Soto et al. (2016) were corrected to consider only the first 30 cm of the soil. This correction factor was set at 61%, which is the average of temperate coniferous forest, temperate deciduous forest, and sclerophyllous shrubs in Jackson et al. (1996).

Total cropland area represents the area actually cropped, including temporary fallow. SOC_{eq} in the total cropland area is proportional to cropland SOC concentration, which can be considered a proxy of soil quality and adaptation potential (Lal et al., 2011; Aguilera et al., 2013). A given configuration of land uses and practices resulting in high equilibrium SOC (SOC_{eq}) in the total studied area but low SOC_{eq} in the total cropland area would be valuable for GHG mitigation (high total C stored) but not for long-term productivity and adaptability (low SOC in cropland soils).

It was assumed that SOC was at equilibrium at the starting point (1900). The model was first run up to the second time step (1910), and before initiating a new cycle, the land use changes that had taken place during the decade were applied by pooling together all land use types losing area, and correspondingly allocating their area to the land use types gaining area. This process was iterated in subsequent cycles.

2.8. Sensitivity analysis

The simulations were subjected to a sensitivity analysis to test the effect of changing key variable parameters on the results. The tested parameters were root biomass (two scenarios: Fixed root-shoot and Fixed root mass), weed biomass (one scenario: No weeds), manure (Manure h) and climate (Fixed climate). The scenarios are described in Table A.7.

3. Results

3.1. Net primary production

Cropland NPP reached a first peak in 1933 and fell after the Spanish Civil War (1936–1939), reaching its minimum in 1950 at 2.5 Mg C ha^{-1} yr^{-1}, when it started a steady growth, although it did not surpass the 1933 level until 1980 (Fig. 4a). Overall, cropland NPP, averaged across all crops, grew 37% during the studied period, up to 3.5 Mg C ha^{-1} yr^{-1} in 2008. NPP grew 13% in rainfed systems and 34% in irrigated systems (Fig. 4b, c). NPP of irrigated systems doubled that of rainfed systems in 1900, and tripled it in 1980. NPP growth in herbaceous crops was less pronounced in rainfed (30%) than in irrigated systems (60%), while it was negative both in rainfed (−40%) and irrigated (−16%) woody crops (Fig. 4e, f, h, i).

Crop production was the NPP component with the largest increase, ranging from 61% in rainfed and irrigated woody crops (Fig. 4h, i), to 220% and 146% in rainfed and irrigated herbaceous crops, respectively (Fig. 4e, f). The growth in total NPP considering the total area of each crop type was higher, of 99% and 277% in woody and herbaceous crops, respectively, reflecting the effect of the expansion of irrigation (Fig. 2).

The share of crop residues in total cropland NPP grew from 23% in 1900 to 30% in 2008, despite the decline in harvest indices. Used residues represented more than two-thirds of the extracted biomass in 1900, falling to about one quarter in 2008. Meanwhile, the share of residues burnt in the field grew from 1% in 1900 to 21% in 1990, dropping to 11% in 2008.

Root:shoot ratios of most crop types declined (Table A.8), while root biomass production per hectare increased (Table A.9) during the studied period, which is related to shoot productivity increases.

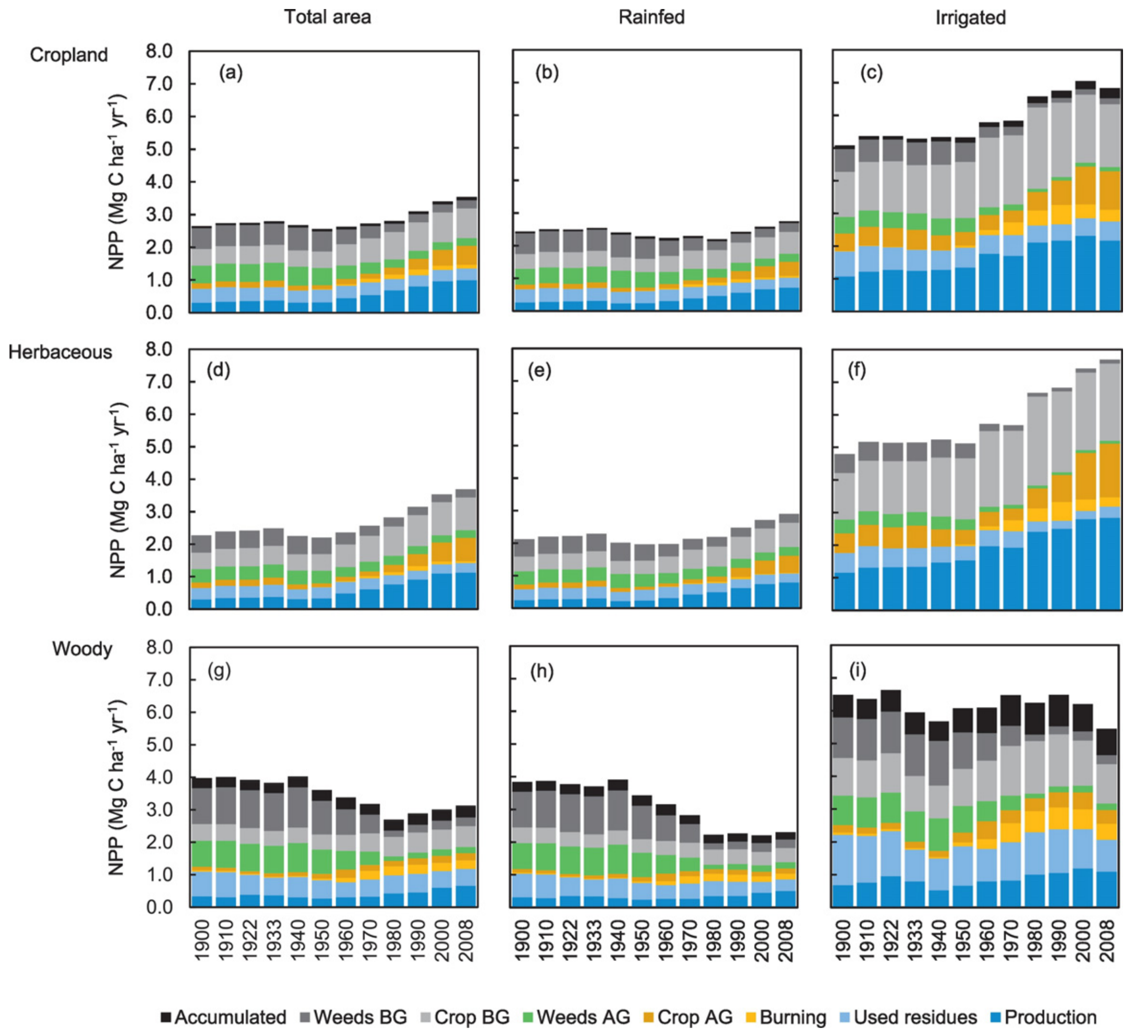

Fig. 4. Historical evolution of Net Primary Production (NPP) in Spanish total (a), rainfed (b) and irrigated (c) cropland, total (d), rainfed (e) and irrigated (f) herbaceous crops, and total (g), rainfed (h) and irrigated (i) woody crops. NPP components are categorized according to their source (crop or weed), their placement (aboveground, *AG*, and belowground, *BG*), and their final use.

Weed biomass production averaged 1.1 Mg C ha^{-1} yr^{-1} during the first half of the 20th century, falling to 0.5 Mg C ha^{-1} yr^{-1} from 1980 onward (Fig. 4a). The drop was specially marked for woody crops (Fig. 4g), from 1.9 to 0.5 Mg C ha^{-1} yr^{-1}, accounting for 48% and 15% of the NPP of woody cropping systems, respectively.

3.2. Carbon inputs to the soil

Average soil C inputs in the total cropland area grew from 2 to 2.4 Mg C ha^{-1} yr^{-1} during the studied period, with a significant drop down to 1.8 Mg C ha^{-1} yr^{-1} in the 1970s (Fig. 5a). This decline was first associated to declining crop productivity and NPP during the 1933–1950 period, and from 1950 to 1980 due to the drop in annual weed biomass production (Fig. 4). The subsequent increase from 1980 (Fig. 5a) can be largely explained by increasing crop residue application due to a combination of an overall increase in crop productivity and a decrease in the relative use and burning of crop residues (Fig. 4).

Overall, soil C inputs in rainfed systems were similar in 1900 and 2008, at 1.9 Mg C ha^{-1} yr^{-1} after having fallen 27% in 1980 from their 1933 peak at 2.0 Mg C ha^{-1} yr^{-1} (Fig. 5b). Soil C inputs in irrigated systems peaked in 2000 at 4.3 Mg C ha^{-1} yr^{-1}, up 20% from 1900 levels (Fig. 5c).

Crop residues applied to the soil reached a first peak (0.2 Mg C ha^{-1} yr^{-1}) in 1933, just before Spanish Civil War, and declined to 0.1 Mg C ha^{-1} yr^{-1} by 1950, but they started growing strongly after 1980, reaching 0.6 Mg C ha^{-1} yr^{-1} in 2008 (Fig. 5a). The contribution of crop residues to total soil C inputs was lowest for woody crops in older time steps, ranging 3–6% (Fig. 5h, i), and highest for herbaceous crops in modern time steps, ranging 21–31% (Fig. 5e, f).

Crop C allocated belowground represented a very significant fraction of soil C inputs. On average, their relative contribution rose from 26% in 1900 to 47% in 1980. Total belowground contribution to soil C inputs, including weeds, ranged 50–70% across all crop-management categories and periods (Fig. 5).

Weed biomass played a major role in the observed trends in soil C inputs. Until 1960, it represented about 50% and 75% of soil C inputs in herbaceous and woody crops, respectively (Fig. 5d, g), whereas by 2008 it had fallen to 18% and 30%, respectively. Weed C had a greater relevance in rainfed than in irrigated crops (Fig. 5e, f, h, i). This trend in weed biomass largely explains the decrease in soil C inputs during the 1950–1980 period in a context of strong growth in crop productivity (Fig. 4), as well as the diverging trends in soil C inputs in woody and herbaceous crops (Fig. 5d, g).

Manure C inputs, which include the deposition of animals grazing in cropland, rose from 0.2 to 0.4 Mg C ha^{-1} yr^{-1} (Fig. 5a) along the studied period, but their contribution to total soil C inputs was always relatively low, ranging 8–16%. The role of urban waste in the C balance was always marginal, barely reaching 1% of soil C inputs in 2008.

Overall, the average changes in soil C inputs (Fig. 5a) can be largely explained by the net balance between the increase in C inputs due to

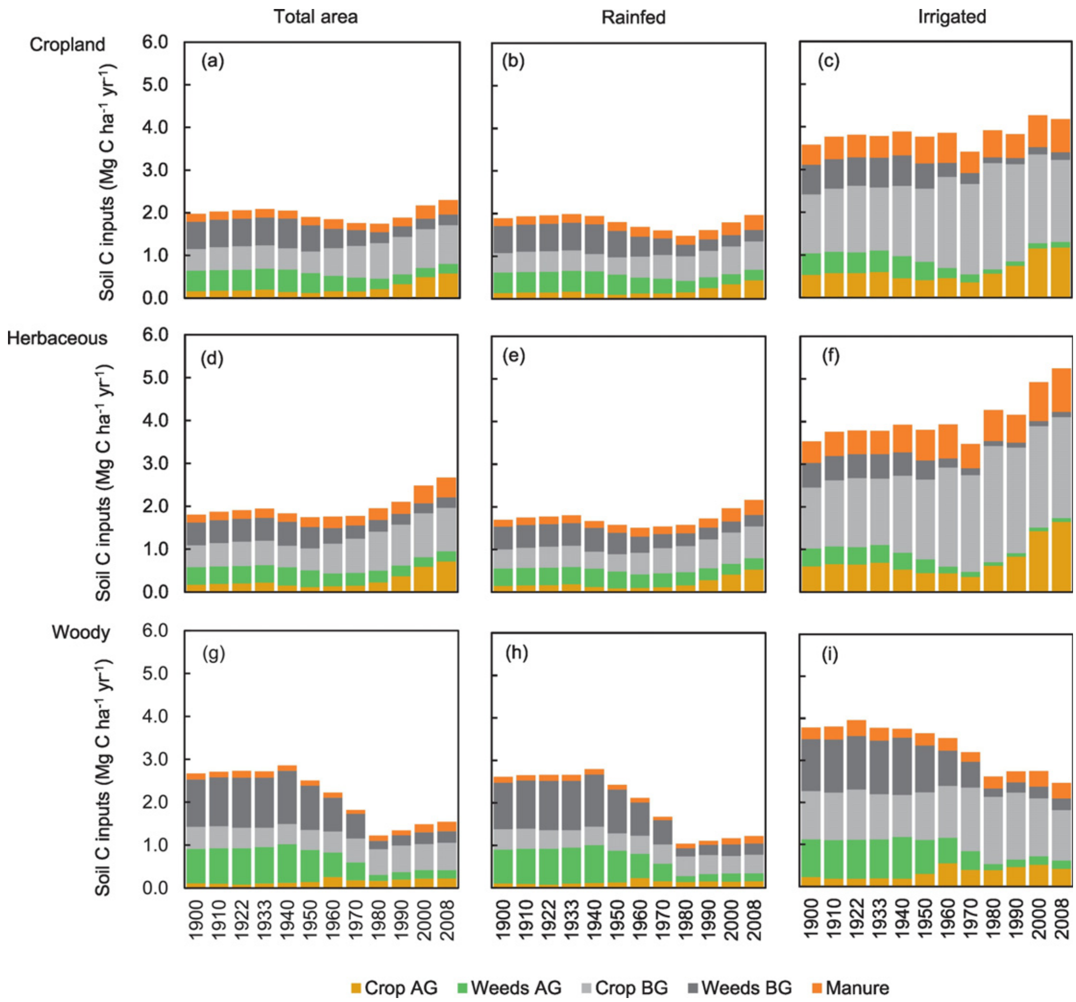

Fig. 5. Historical evolution of soil carbon (C) inputs in Spanish total (a), rainfed (b) and irrigated (c) cropland, total (d), rainfed (e) and irrigated (f) herbaceous crops, and total (g), rainfed (h) and irrigated (i) woody crops. Soil C inputs components are categorized according to their source (crop, weed or external), and their placement (aboveground, *AG*, and belowground, *BG*). Manure category includes manure and urban waste.

the expansion of irrigation, and the decrease in C inputs due to the expansion of woody crops (Fig. 2), in which inputs decreased strongly from 1940 to 1980 due to the loss of soil cover (Fig. 5g). The analysis by crop types (Table A.9) also reveals strong differences in soil C inputs among the different crops, which contributed to the average changes observed. For example, highest soil C inputs were observed for alfalfa, and the expansion of this crop contributed to the growth in total cropland soil C inputs after 1950.

3.3. SOC decomposition rates

The changes in management and climate exerted strong effects on SOC decomposition rates, which varied depending on the geographical distribution and specific agronomic features of each crop-management category (Table 3). In rainfed herbaceous crops, with no management changes affecting decomposition rates, HUM decomposition rate increased by 21–34%, with lowest rates for winter cereal crops, cultivated mainly in the central provinces of the country (Fig. C.1), with a continental Mediterranean climate, and highest rates for summer cereals, cultivated mainly in the Northern provinces (Fig. C.2), with a temperate climate. The increase in decomposition rates was higher for woody crops, in which the effect of soil cover loss was added to the effect of the rise in mean annual temperature.

Decomposition rates under irrigation were higher than in rainfed systems (Table 3), but these differences were much lower for winter cereals (56% in 1900), which are commonly irrigated only in spring, than for summer cereals (108% in 1900), which are irrigated specially during the period of high temperatures and with higher water doses. Differences between rainfed and irrigated systems were even higher (above 200% in 1900) for other herbaceous crops and treenuts, olive and vineyards, which are broadly cultivated in the coastal Mediterranean and Southern provinces, with a warmer climate. The trends in decomposition rates estimated for irrigated crops were very heterogeneous, increasing by 17–29% in cereals and forage crops, mainly due to increasing temperatures, but declining by 7–19% in other herbaceous crops, citrus, olive and grapevine. The observed decline took place during the last two decades, and it was mainly related to the expansion of drip irrigation, which favors water deficit conditions limiting SOC decomposition.

3.4. SOC simulations

The simulated maximum SOC level in the total studied area was 1.19 Tg C in 1900, declining throughout the 20th century, with a minimum in 2000 at 0.96 Tg C (Fig. 6a). The estimated equilibrium SOC reached a minimum at 0.74 Tg C in 1980, and in 2008 it was still 18% lower than in 1900 (Fig. 6b).

In the general analysis of the cultivated cropland area of Spain (Fig. 7a), there was a first period of growth in SOC from 1900 to 1933, partially associated to the increase in soil C inputs, but mainly to the expansion of cropland over non-cropland. This is, part of the increase was only apparent, as the new cropland area included previously non-cropped land with a higher SOC content. Cropland SOC fell steadily

Table 3
Annual C mineralization rates, K_2 (% SOC yr^{-1}) of HUM (slow turnover SOC pool) in simulated crop and land use categories in Spain along the studied period.

	1900–1909	1910–1921	1922–1932	1933–1939	1940–1949	1950–1959	1960–1969	1970–1979	1980–1989	1990–1999	2000–2008
Rainfed											
Winter cereals	0.73	0.75	0.80	0.84	0.80	0.79	0.87	0.77	0.84	0.86	0.97
Summer cereals	1.14	1.20	1.33	1.31	1.39	1.30	1.43	1.30	1.36	1.60	1.49
Forage	0.89	0.92	1.00	0.99	1.02	0.98	1.05	0.96	1.00	1.10	1.08
Other herbaceous	0.87	0.88	0.95	0.96	0.95	0.92	0.99	0.88	0.97	0.96	1.09
Fruits	0.87	0.89	0.92	0.95	0.93	0.99	1.11	1.07	1.20	1.27	1.39
Treenuts	0.77	0.78	0.81	0.80	0.78	0.80	1.01	1.00	1.09	1.03	1.17
Olive	0.87	0.86	0.92	0.91	0.87	0.90	1.06	0.97	1.15	1.08	1.26
Grapevine	0.71	0.72	0.76	0.77	0.73	0.74	0.86	0.79	0.97	0.94	1.09
Fallow	0.89	0.89	0.95	0.97	0.93	0.92	1.01	0.91	0.98	0.97	1.08
Non-cropland	0.67	0.67	0.72	0.73	0.69	0.69	0.76	0.68	0.73	0.72	0.78
Irrigated											
Winter cereals	1.13	1.14	1.18	1.21	1.22	1.20	1.29	1.15	1.28	1.36	1.45
Summer cereals	1.97	1.99	2.04	2.04	2.14	2.13	2.13	2.04	2.12	2.11	2.32
Forage	1.95	1.95	1.98	1.99	2.07	2.09	2.09	2.01	2.17	2.24	2.29
Other herbaceous	2.68	2.67	2.78	2.76	2.92	2.84	2.84	2.70	2.77	2.58	2.26
Fruits	2.29	2.29	2.32	2.35	2.35	2.53	2.63	2.63	2.98	2.90	2.54
Citrus	2.67	2.65	2.71	2.69	2.80	2.88	3.06	3.06	3.49	3.24	2.49
Treenuts	2.59	2.58	2.65	2.62	2.70	2.80	3.00	3.19	3.38	3.22	2.54
Olive	2.69	2.69	2.80	2.77	2.90	2.91	3.12	3.11	3.49	3.21	2.19
Grapevine	2.38	2.38	2.43	2.42	2.51	2.55	2.62	2.59	3.07	2.89	2.22

after this peak, reaching its minimum at 45.1 Mg C ha^{-1} by 2008. Fig. 7a also reveals the contrasting trends between herbaceous and woody crops: while SOC stocks declined 12% from 1933 to 2008 in herbaceous crops, the decrease was 37% for woody crops. The observed SOC patterns for rainfed crops (Fig. 7b) were similar to the general trends in cropland, but they were markedly different for irrigated crops

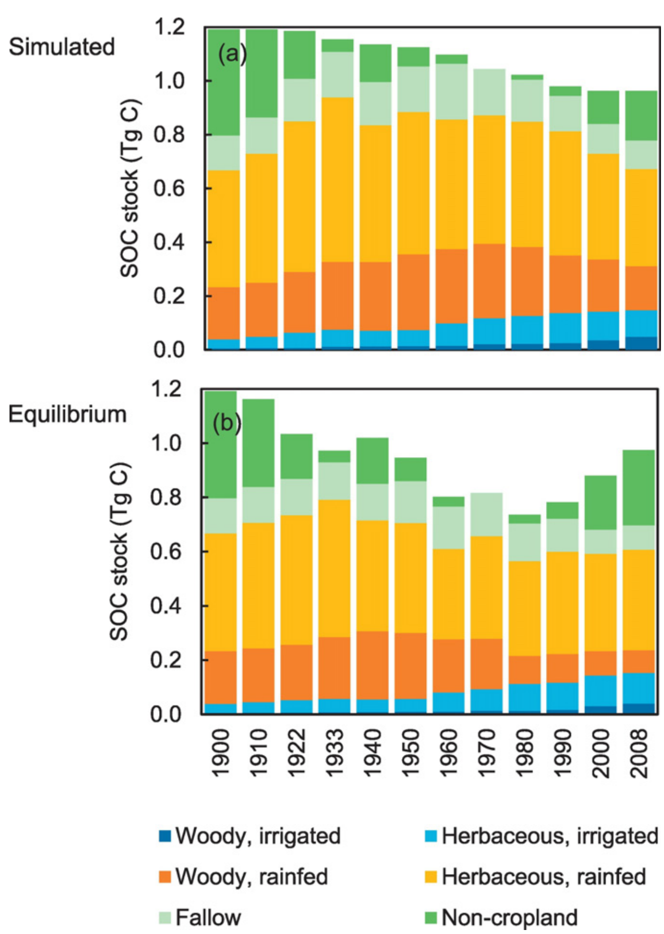

Fig. 6. Simulated total SOC stock (a) and SOC stock at equilibrium (b) at 30 cm depth for the total studied area.

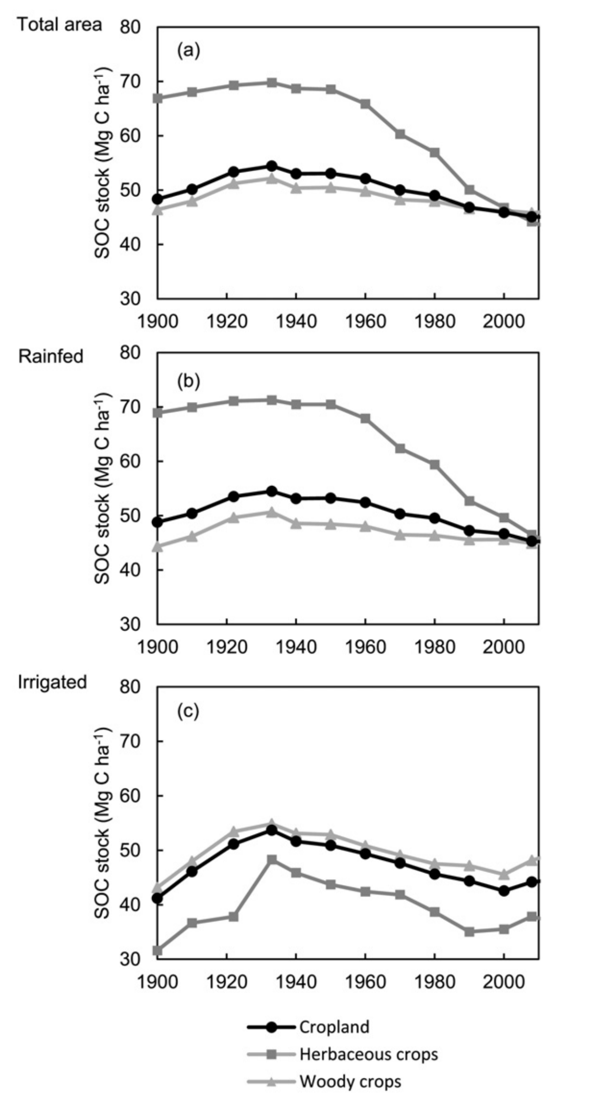

Fig. 7. Simulated SOC stocks (at 30 cm depth) per hectare for total (a), rainfed (b), and irrigated (c) cropland in Spain.

(Fig. 7c). Under irrigation, SOC levels started recovering by 2000, and they were lower for woody crops than for herbaceous crops.

3.5. Sensitivity analysis

Most of the scenarios in the sensitivity analysis showed SOC values similar to those of the base scenario. The root biomass calculation method (Fig. 8a–d) had a very limited influence on SOC values, with slightly lower estimates for the fixed root-shoot approach, and higher for the fixed root mass approach. The observed decline in SOC from 1900 to 2008 in the total studied area was similar for the fixed root:shoot approach and for the fixed root mass approach (Fig. 8a). The differences between the crop root estimation approaches were higher for herbaceous crops (Fig. 8c) than for woody crops (Fig. 8d), reflecting the lower share of crop roots in the NPP of woody crops.

Applying a manure humification coefficient of 12% hardly affected SOC stocks, leading to just 2–8% lower SOC estimates than in the Base scenario (Fig. 8e–h), with an estimated total SOC decline 5% higher than in the Base scenario.

Fixed climate parameters instead of decadal averages led to 10% lower total SOC stock at the beginning of the 20th century, a difference that was progressively reduced up to 2% in 2008 (Fig. 8e). This implies that temperature increase was responsible for the loss of ca. 4.8 Mg C ha^{-1} in the total studied area during the 1900–2008 period. For the whole period, the average rate of decline due to changes in climate was 0.04 Mg CO_2e ha^{-1} yr^{-1}, but it progressively increased from 0 in the 1900–1910 period to 0.13 Mg CO_2e ha^{-1} yr^{-1} in the 2000–2008 period.

Weed biomass was the factor with the highest influence on SOC estimates (Fig. 8e–h). Despite a reduction in SOC in the total area is also observed when weeds were excluded (Fig. 8e), the total amount of SOC stored was less than half of that of the Base scenario at the beginning of the study period. This is mainly due to the major contribution of weeds to total C inputs of traditional cropping systems (Fig. 5). SOC remained stable for cropland in the scenario without weeds in the 1940–2008 period, a markedly different pattern compared with all the other scenarios.

4. Discussion

This study depicts the evolution of SOC stock in Spanish cropland as a complex story involving shifting trends driven by changes in land uses, management and climate. In turn, these changes are determined by historical events in Spain and the world. The results shed light on the implications of the transition from an organic to an industrial socio-metabolic regime on SOC dynamics, and on the potential and limitations of traditional organic management practices to increase SOC storage in modern Mediterranean agroecosystems.

4.1. Historical periods associated to the observed trends

Four distinct historical periods help explaining the observed trends in SOC stocks. In the first period (1900–1936), both cropland area expansion and intensification took place within the traditional organic agriculture model. The result was a decline in SOC stocks considering the total studied area (Fig. 6a), as the expansion of cropland dominated over the increase in cropland soil C inputs (Fig. 5) in a context of stable SOC decomposition rates (Table 3).

The second period (1936–1955), was characterized by low yields (Fig. 4) and cropland abandonment (Fig. 2, Soto et al. 2016) due to the Spanish Civil War and Franco's dictatorship autarky period. In this period, SOC stocks further declined because of a decrease in cropland NPP (Fig. 4) and soil C inputs (Fig. 5), which prevailed over cropland abandonment (Fig. 6).

During the third period (1955–1986), Franco's regime transitioned from autarky to liberal economic policies, combined with investment in irrigation infrastructure, promoting a rapid adoption of industrial inputs. This policy was further developed after the onset of democracy in 1978. This was the most intensive stage of the industrialization process, in which external energy inputs increased 11-fold (Guzmán et al., 2017). This period shows the highest SOC decrease rate (Fig. 6), which was mainly related to declining soil C inputs, particularly from weeds (Fig. 4), combined with increasing SOC decomposition rates (Table 3) due to irrigation expansion and soil weed-cover loss in woody crops. The decline in cropland SOC observed after the mid-20th century is in

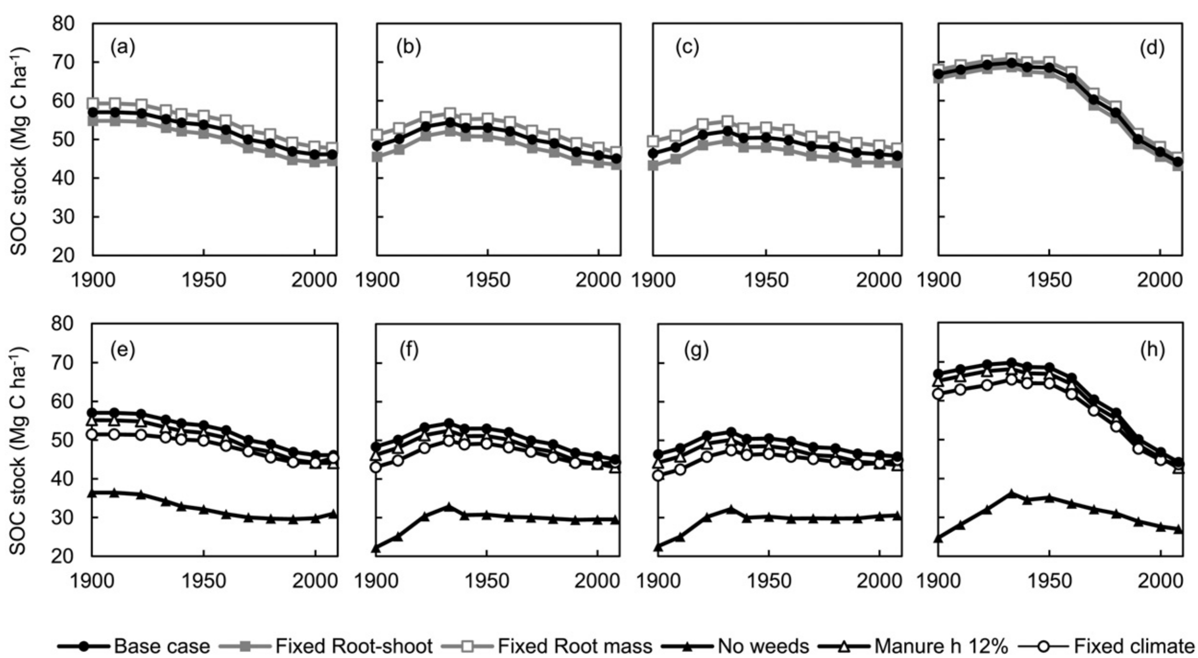

Fig. 8. Sensitivity analysis of some key methodological choices affecting SOC in the total studied area of Spain (a, e), total cropland (b, f), herbaceous crops (c, g) and woody crops (d, h). Studied scenarios include the base case (the simulation performed in the rest of the paper), Fixed root-shoot (fixed crop-specific root:shoot ratios for the calculation of root biomass), Fixed root mass (Fixed crop-specific root biomass values), No weeds (weeds are excluded from the model), Manure h (manure humification coefficient is changed), and Fixed climate (climate parameters are maintained constant along the simulation period).

line with field observations in England and Wales from 1978 to 2003 (Bellamy et al., 2005) or Belgium from 1955 to 2005 (Goidts et al., 2009), and with modeling studies in Europe from 1951 to 1965 (Ciais et al., 2011). In Belgium, changes in the agricultural practices and in the precipitation regime were identified as the major causes of cropland SOC loss (Goidts et al., 2009). In the European simulation, the decrease was mainly linked to the shifting from organic to mineral fertilizers (Ciais et al., 2011). In the US Great Plains, the soil shifted from being a CO_2 source to a C sink during this period (Evrendilek and Wali, 2001; Parton et al., 2015), mainly due to the increase in residue production and soil application.

The last period (1986–2008) started with the entrance of Spain in the European Union, strengthening the market orientation of its agricultural sector but also incorporating environmental policies, such as restriction of residue burning. This period was also characterized by the relative abandonment of cereals straw use for animal feeding and bedding, supported by imported biomass (Soto et al., 2016). SOC stocks at equilibrium, and ultimately simulated SOC, progressively increased (Fig. 6), mainly driven by: (i) growing crop residue soil C inputs (Fig. 5), due to a combination of increasing residue production, decreasing residue field burning, and a stabilization of residue use, and (ii) diverging trajectories in SOC decomposition rates (Table 3), which grew in some areas due to rising temperatures, but decreased in other due to the expansion of drip irrigation and cropland abandonment. This land sparing process improved equilibrium SOC levels compared to those of the 1970s, but not to those of the early 20th century. The observed non-linear trend in SOC stocks suggest that it is not intensification per se which drives SOC changes, but the way in which this intensification is implemented. In Spain, as in other areas (e.g. Firbank et al., 2013), agricultural intensification is currently getting more sustainable, but there is still a very large potential for improvement (González de Molina and Guzmán Casado, 2017).

4.2. Comparison of simulated SOC with field studies

We have found no historical data to validate our simulations of SOC trends in Spanish cropland. However, there are some published estimations of SOC contents in the last decades that can be used for comparison. Rodríguez-Murillo (2001) estimated SOC contents in Spanish cropland based on soil surveys and found slightly higher SOC contents in irrigated than in rainfed areas, similar to our simulation around 1980. However, the variability in the data was very high, probably due to the large range of soil depths (up to 1 m) and time points (1960–1995) included, hindering the comparability with our results. In another field assessment of Spanish soils, Romanya and Rovira (2011) found higher SOC values for irrigated cereals than for rainfed cereals in Mediterranean alkaline soils, a similar pattern than the one observed in this study for the whole cropland area in 2008. In the case of olives and nuts, Romanya and Rovira (2011) observed similar SOC contents for both categories, whereas we found lower SOC values under irrigation than in rainfed areas. This may indicate an under-estimation of SOC in irrigated woody crops in our assessment, probably related to an over-estimation of water inputs in these systems.

The most appropriate comparison of our results with measured values can be done using the comprehensive assessment of Rodríguez-Martín et al. (2016). Simulated SOC levels in year 2008 were 0.3 Mg C ha^{-1} lower for herbaceous crops and 5.3 Mg ha^{-1} higher for woody crops than the average values estimated from field measurements. Both simulated data, however, were within the confidence intervals of the data in Rodríguez-Martín et al. (2016).

Equilibrium SOC values modeled for non-cropland, ranging 69–90 Mg C ha^{-1} (data not shown), were within the range of values reported by Rodríguez-Martín et al. (2016) and Doblas-Miranda et al. (2013), based on field measurements. Because of these relatively high SOC values, there was a strong response of SOC to abandonment, which has also been observed in experimental studies in Spain (Segura et al., 2016).

4.3. Factors affecting the observed changes in NPP, soil C inputs and SOC

4.3.1. General considerations

The limited growth in NPP can be partially attributed to climate constraints, but it has also been linked to the degradation of the agroecosystem functions caused by industrialization (Guzmán et al., 2017), which may include the observed decline in SOC and the effect of anthropogenic climate change. Ciais et al. (2010) found that rainfall changes consisting on dryer springs and wetter autumns contributed to C losses in western Mediterranean regions. These authors also estimated that atmospheric CO_2 increase was responsible for a 10% NPP increase in European arable lands, indicating that a large part of the NPP increase that we observed could be attributed to CO_2 enrichment. Another important factor in the NPP and SOC trends was cover crop loss in woody cropping systems. The situation observed in Spain contrasts with that of Italy, where cover crops in woody crops have not declined. Farina et al. (2017) found higher SOC values for woody crops than for herbaceous crops in Southern Italy, which was associated to the high contribution of cover crops to soil C inputs, as we observed for the first half of the 20th century in Spain.

The small disagreement with measured data from Rodriguez-Martín et al. (2016) for woody crops could be due to a number of factors not accounted for in the model, such as previous management history, soil drying and wetting cycles, erosion, increases in atmospheric CO_2, or management practices such as tillage. Drying and wetting cycles are an important factor controlling SOC dynamics in Mediterranean environments (Jarvis et al., 2007), but they may have a limited effect on annual SOC changes (Borken and Matzner, 2009). Soil C loss by erosion in Mediterranean woody cropping systems was enhanced by a factor of 2–40 by the absence of soil cover (Gomez et al., 2011), which can result in a loss of 1–12 Mg C ha^{-1} over a 50-year period. Vanwalleghem et al. (2011) found that soil loss increased at least 3-fold from 1900 to 1970 to 1970–2000 in olive groves in Southern Spain. The strong increase in mechanical traction in the 1950–1980 period (Guzmán et al., 2017) was probably related to an increase in tillage intensity and depth. After 1990, conservation tillage practices expanded up to >3 Mha in 2010, possibly promoting SOC storage in these areas (González-Sánchez et al., 2012). However, the net effects of these changes on SOC is highly uncertain, as they could be offset by changes in soil layers below the 30 cm depth limit of our model.

4.3.2. Irrigation

The results stress the key role of irrigation in the productivity of Mediterranean systems (Wriedt et al., 2009), following the global trend (Ozdogan, 2011), but also in soil C inputs and SOC decomposition rates, both of which roughly doubled with irrigation (Fig. 5, Table 3). These opposing effects resulted in mixed effects on SOC, in line with studies reporting either a decrease (Condron et al., 2014; Nunes et al., 2007) or an increase (e.g. Montanaro et al., 2009) in SOC associated to irrigation. Additionally, the results suggest that drip irrigation was responsible for a substantial drop in C mineralization rates, due to water limitation. Drip irrigation, however, could also reduce the root:shoot ratio when compared to furrow irrigation (Araujo et al., 1995). Therefore, there is a need for experimental studies specifically addressing the net effect of drip irrigation on SOC, especially given its interaction with other GHG (Cayuela et al., 2017; Sanz-Cobena et al., 2017). Irrigation should also be assessed considering its broader impacts on the water cycle, as its expansion has severely affected freshwater ecosystems in most Mediterranean and semiarid basins (Romero et al., 2016; Jaramillo and Destouni, 2015) and may lead to changes in rainfall regimes (Lo and Famiglietti, 2013).

4.3.3. Weeds

The outcomes of the simulations and the sensitivity analysis reveal the prominent role of weed biomass in the changes in cropland NPP and SOC related to agricultural industrialization. The reduction in weeds due to herbicide expansion was offset by crop biomass growth in the case of herbaceous crops, but not of woody crops. Cover crops in woody cropping systems have been widely acknowledged as an effective soil C-building practice under Mediterranean conditions (Gonzalez-Sanchez et al., 2012; Aguilera et al., 2013; Vicente-Vicente et al., 2016). Furthermore, decreases in weed biomass production have been associated to lower biodiversity of weeds (Alignier et al., 2017) and wild heterotrophs (Rundlöf et al., 2008; Gabriel et al., 2013).

4.3.4. Crop residues

The increase in harvest indices was another important factor of the initial reduction in cropland soil C inputs during Spanish agriculture industrialization. Sinclair (1998) argued that low harvest indices in traditional systems could allow for higher biomass production under low N availability, being functional in a context in which straw had a high value. Currently, the search for renewable energy sources and soil C sequestration has made straw production desirable again (e.g. Lorenz et al., 2010). High yields can indirectly reduce GHG emissions and other environmental impacts (e.g. Bennetzen et al., 2016), so breeding efforts should focus on dual-purposes varieties characterized by a high residue production without decreasing crop product yield (Lorenz et al., 2010). The focus on residue production may make even more sense in a context of stagnating yields in Europe (Wiesmeier et al., 2015). In Mediterranean organic farming systems, traditional crop varieties could be able to perform as well as modern varieties in terms of grain yield (Carranza et al., 2017), although this has not been observed under temperate climate (Konvalina et al., 2014).

4.3.5. Roots

The estimated root biomass production values and root:shoot ratios of the major herbaceous crops are within the ranges of the global revision by Mathew et al. (2017). Roots represented the majority of C inputs across most of the crop-management categories and time-periods, emphasizing the need to include roots in cropland C assessments. In a global change context, and particularly in low-productivity environments such as many Mediterranean systems, promoting root growth may help simultaneously increasing yields and SOC storage (Paustian et al., 2016), while improving water use (Bodner et al., 2015).

4.3.6. Manure

The contribution of manure to soil C inputs was very limited even in a situation of strong growth of the livestock herd (Soto et al., 2016), which by 2007 consumed half of the feed protein from abroad (Lassaletta et al., 2014). This is related to the decrease in manure C:N ratios in modern livestock farming, even if changes due to animal diet composition, which could be significant (Yamulki, 2006) were not considered.

4.3.7. Climate

Temperature rise gradually became the main driver explaining the observed decrease in SOC, representing a positive feedback to climate change in this Mediterranean area. The negative impact of climate change on SOC levels is particularly worrisome given the low SOC content of Mediterranean soils. Modeling studies suggest that yields could also be negatively affected by climate change in Mediterranean areas (Bindi and Olesen, 2011), and accelerate phenology driven by warming could decrease NPP (Ventrella et al., 2012), which would further reduce C availability for soil application. These trends warn us about the need for substantial changes in management practices in order to maintain soil quality in a global warming context.

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