Universidad de Valladolid

eSDUVa Escuela de Doctorado Universidad de Valladolid

PROGRAMA DE DOCTORADO EN CIENCIA E INGENIERÍA AGROALIMENTARIA Y DE BIOSISYEMAS

TESIS DOCTORAL:

Effect of tillage systems on soil properties, water dynamics and greenhouse gas emissions in a continuous irrigated maize crop in semi-arid conditions.

Presentada por Maroua Dachraoui para optar al grado de Doctora por la Universidad de Valladolid

Directora:

Dra. Aurora Sombrero Sacristán

Tutora:

Dra. Mercedes Sánchez Bascones

to mom

This work was funded by the projects (RTA2010-00006-C03-01 and RTA2013-00009-C02-02) of The National Institute for Agricultural and Food Research and Technology (INIA).

AGRADECIMIENTOS

En primer lugar, me gustaría agradecer al Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria (INIA) por la financiación de mi estancia y al Instituto Tecnológico de Castilla y León (ITACyL) por poner a mi disposición sus instalaciones y medios técnicos para la realización del trabajado.

Un especial agradecimiento a mi directora de tesis Dra. Aurora Sombrero Sacristán, con quien he trabajado cada día y de la cual he aprendido tanto. Aurora, te agradezco mucho mi acogida, tus consejos, tu contribución en mi progreso a nivel tanto profesional como personal.

De igual manera, gracias a Dra. Mercedes Sánchez Bascones, Dr. Salvador Hernández Navarro y Dr. Luis Manuel Navas Gracia por sus apoyos y ayudas cada vez que lo necesitaba.

Mis más sinceros agradecimientos a mis compañeros de trabajo Arturo, Alicia, Berta, Rodrigo, Yolanda, Bea, Ismael, Lidio, Carmelo, Yesica, etc.… y a todos aquellos quien han estado presentes a lo largo de estos cuatro años. Ya no os considero como mis compañeros, pero como la familia que me faltaba aquí. Me habéis enseñado mucho y habéis aguantado mis caprichos con una sonrisa, os doy las gracias. Sin olvidar a Marie Carmen quien me dio el último empujón para acabar este trabajo y María Ángeles para aguantar mi ignorancia administrativa.

A Mila y Pablo, me habéis salvado el pellejo con tanto papeleo.

A toda mi familia, con especial mención a mi abuela, mi madre y mi hermano a quien debo todo el éxito que he realizado, por su continuo apoyo y por su paciencia, os echo de menos. A Kaouther y Fatma por creer en mí.

A Mario, por todo el cariño y el amor que me ofrece cada día sin pedir nada a cambio, sabes cuánto te quiero, sino no estarás en este apartado.

“El que se porta gentil y cortésmente con los extranjeros, demuestra ser ciudadano del mundo.”

SIR FRANCIS BACON

Gracias a todos por hacerme sentir como en casa.

TABLE OF CONTENTS

INDEX OF TABLES

INDEX OF FIGURES

ABSTRACT

The use of conventional tillage (CT) for continuous maize crop (*Zea mays*) is widely practiced by local farmers in Northwest Spain. This includes a frequent soil ploughing and disturbance to prepare the seedbed and to control weeds before sowing. Conversion from CT to no-tillage (NT) would not only help increase soil organic carbon (SOC), reduce soil erosion and CO_2 emissions but also can improve soil water content (SWC), lower the carbon footprint (CFP) and environmental impacts. For these reasons it was interesting to evaluate the effect of CT and NT on various aspects of the production cycle of a continuous irrigated maize crop in Castile and Leon.

Therefore, this study was carried out from 2011 to 2017 in Zamadueñas's experimental field, located in the Spanish province of Valladolid. The experimental design included four random blocks where the main factor studied was tillage systems. Under CT, the seedbed was prepared with a mouldboard plough followed by a spring cultivator, while only an herbicide (glyphosate) was applied under NT system for weeds control. The different soil properties were determined by collecting soil samples up to 100 cm soil depth after the maize harvest in 2011, 2015 and 2017. The SWC was monitored during the maize production cycle and CO_2 emissions were measured at different intervals from 2011 to 2017. Finally, the assessment of the carbon footprint (CFP) included the agricultural inputs of the maize production process, the N_2O emissions which were estimated using the methodology suggested by IPCC (2006) and SOC changes.

The evaluation of the soil properties revealed that some soil parameters such as bulk density, pH, available phosphorus and extractable potassium could not only be affected by tillage systems but also by the combination of the experiment conditions and the climatic variations from year to year. Nevertheless, the soil organic matter was significantly higher under NT system than CT thanks to the addition of crop residues and the non-disturbance of the soil surface.

The results showed that NT treatment presented higher SWC especially in the first 50 cm and 100 cm soil depth than CT system during the 3-year study. Tillage system did not show significant differences in the assessment of water productivity. However, the mean water use efficiency and irrigation use efficiency were 4.0 and 4.5% higher under NT system than CT system.

In 2017, SOC stock was 36% greater under NT than CT, with a rate of 1.6 and 1.1 Mg ha^{-1} yr^{-1} respectively at 0-10 cm depth. The SOC stock of the top 30 cm soil layer was 13% greater under NT system than CT. CO_2 emissions were significantly affected by tillage systems in short and long terms and were significantly higher under CT than NT system. Moreover, the results obtained showed that the emissions from the agricultural inputs of fuel and electricity inputs (direct energy) and machinery use were higher under CT system while the ones emitted from synthetic fertilizers, pesticides, water applications and maize seed (indirect energy) were greater under NT treatment. The highest N_2O emissions were produced by the application of the highest rates of N fertilization.

The absence of soil disturbance combined with crop residue retention increased the SOC accumulation in the topsoil layer, promoted the accumulation of moisture in the soil during drought period, reduced CO_2 emissions without drastically decreasing maize grain yield and helped minimizing the carbon footprint and the impact on climate change.

Resumen

El uso del laboreo convencional (CT) en el cultivo de maíz (*Zea mays*) es ampliamente practicado por los agricultores locales del noroeste de España. Esto incluye un arado y volteo frecuentes del suelo para preparar el lecho de siembra y para controlar las malas hierbas antes de sembrar. La conversión del LC al no laboreo (NT) no sólo ayudaría a aumentar el carbono orgánico del suelo (SOC), reducir la erosión del suelo y las emisiones de CO_2, sino que también puede mejorar el contenido de agua del suelo (SWC), reducir la huella de carbono (CFP) y los impactos ambientales. Por estas razones ha sido interesante evaluar el efecto de los sistemas CT y NT en varios aspectos del ciclo productivo del cultivo de maíz en regadío en Castilla y León.

Por ello, este estudio se llevó a cabo desde 2011 hasta 2017 en el campo experimental de Zamadueñas, situado en la provincia de Valladolid. El diseño experimental incluyó cuatro bloques aleatorios en los que el principal factor estudiado fue el sistema de laboreo. En el sistema CT, el lecho de siembra se preparó con un arado de vertedera seguido de un cultivador de primavera, mientras que en el sistema NT sólo se aplicó un herbicida (glifosato) para el control de las malas hierbas. Las diferentes propiedades del suelo se determinaron mediante la recogida de muestras de suelo hasta 100 cm de profundidad después de la cosecha de maíz en 2011, 2015 y 2017. El SWC fue monitoreado durante el ciclo de producción de maíz y las emisiones de CO_2 se midieron en diferentes intervalos desde 2011 hasta 2017. Por último, la evaluación de la CFP incluyó los insumos agrícolas del proceso de producción de maíz, las emisiones de N_2O que se estimaron utilizando la metodología sugerida por el IPCC (2006) y los cambios de SOC.

La evaluación de las propiedades del suelo reveló que algunos parámetros del suelo, como la densidad aparente, el pH, el fósforo disponible y el potasio extraíble, no sólo podían verse afectados por los sistemas de laboreo, sino también por la combinación de las condiciones en las que se llevó a cabo el experimento y las variaciones climáticas de un año a otro. Sin embargo, la materia orgánica del suelo fue significativamente mayor en el sistema NT que en el CT debido a la adición de residuos de cultivos y a la no alteración de la superficie del suelo.

Los resultados mostraron que el sistema NT presentó mayor SWC especialmente en los primeros 50 cm y 100 cm de profundidad del suelo que el sistema CT durante los 3 años de estudio. El sistema de laboreo no mostró diferencias significativas en la evaluación de

la productividad del agua. Sin embargo, la media de eficiencia del uso del agua y la eficiencia del uso del riego fueron un 4,0 y un 4,5% mayores en el sistema NT que en el sistema CT.

En 2017, el stock de SOC fue un 36% mayor en el sistema NT que en el sistema CT, con una tasa de 1,6 y 1,1 Mg ha^{-1} $año^{-1}$ respectivamente a 0-10 cm de profundidad. El contenido de SOC en los primeros 30 cm de profundidad fue un 13% mayor en el sistema de NT que CT. Las emisiones de CO_2 se vieron significativamente afectadas por los sistemas de laboreo a corto y largo plazos y fueron significativamente mayores en el sistema CT que en el sistema NT. Además, los resultados obtenidos mostraron que las emisiones de los insumos agrícolas de combustible y electricidad (energía directa) y del uso de maquinaria fueron mayores en el sistema CT, mientras que las emitidas por los fertilizantes sintéticos, pesticidas y aplicaciones de agua y semillas de maíz (energía indirecta) fueron mayores en el sistema NT. Las mayores emisiones de N_2O se produjeron por la aplicación de las mayores cantidades de fertilización nitrogenada.

La ausencia del volteo de suelo combinada con la retención de residuos de cultivos aumentó la acumulación de SOC en la superficie del suelo, promovió la acumulación de humedad en el perfil del suelo durante el período de sequía, redujo las emisiones de CO_2 sin disminuir drásticamente el rendimiento de maíz y ayudó a minimizar la huella de carbono y el impacto en el cambio climático.

CHAPTER 1. GENERAL INTRODUCTION

1.1. A brief history of tillage

From as early as 11.000 BC, people began a gradual transition away from a hunter-gatherer lifestyle toward cultivating crops and raising animals for food. The planned cultivation of useful plants probably didn't stem from any conscious desire to create a better society, but was born of necessity, as the high population density meant that hunting grounds were depleted (Harari, 2015).
Besides the increasing density of population, the changes of climate and the outbreak of a wide variety of tillage tools, ranging from a simple digging stick to a paddle-shaped spade or hoe that could be pulled by humans or animals (Lal et al., 2007), allowed people to cultivate lands and grow their own crops. The shift to agriculture is believed to have occurred independently in several parts of the world, including northern China, Central America, and the Fertile Crescent, a region in the Middle East that cradled some of the earliest civilizations (Montgomery, 2007).

Around 4000-6000 BC, it was believed that the soil must be turned to bring nutrients to the surface in order to grow crops. This necessity led to the emergence of a major advance in Mesopotamia known today as the ard (Lal et al., 2007). It consisted of a simple wooden tool that could be strapped to the animal's backs, as they pull along a blade or a wooden stick runs through the soil creating deep gorges and where deeper soil is exposed ready for sowing crops. Over time the ard (Figure 1.1) evolved and additional parts were added to the simple blade such as the share and by the 5th century AD, the plough with iron share was widely used in Europe and the Roman plough evolved into a soil-inverting plough during 8th to 10th century AD (Lerche, 1994).

By 1784, Thomas Jefferson designed the mouldboard plough as we know it today, but it was not until the 1830s that it was manufactured and marketed by a blacksmith named John Deere (Lal et al., 2007). This innovation has been a symbol of U.S. agriculture since about 1850 and allowed farmers to create a soil environment in which grain crops could thrive and meet the needs of the increasing population, but at the same time, it degraded soil structure and promoted crusting, compaction and erosion.

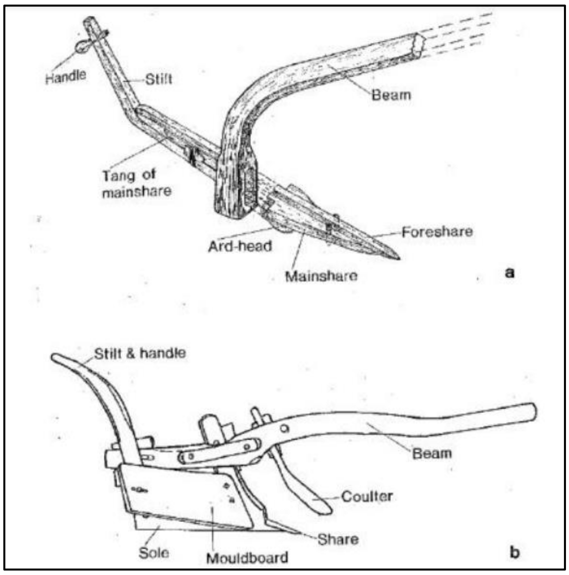

Figure 1.1: The ard (a) and the heavy plough (b) (adapted from Fowler, 2002)

The combination of the soil alteration and prolonged drought had led to severe dust storms in the Great Plains in the USA in the 1930s and the country went through a rough period known as the Dust Bowl (Lal et al., 2007). This event created a controversy about the usefulness of the mouldboard plough and urged the country to develop new techniques to grow crops without excessively disturbing the soil.

By 1943, a book entitled *"Plowman's Folly"* written by Edward Faulkner was published and was considered as a milestone in the history of agricultural practices (Derpsch, 2004). The writer questioned the wisdom of ploughing by stating "No one has ever advanced scientific reason for plowing" (Derpsch, 2004; Lal et al., 2007). Since the 1950s, the agricultural sector has witnessed a gradual transition from the mouldboard plough to decreasing soil disturbance and reducing tillage by using fall or spring chiselling without inverting the soil and by retaining plant residues on the soil surface to alleviate wind and water erosion (Duley and Fenster, 1954; Mannering, 1979). Numerous researchers started to investigate the effects of conservation agriculture (CA) on different parameters of cultivated crops such as grain yield and soil properties (Triplett and Van Doren, 1969; Lal, 1974; Carter and Rennie, 1982).

The excessive tillage of lands has expanded all over the world and the mean rate of soil loss in Europe has been estimated at about 17 mg ha^{-1} yr^{-1}, which exceeds the estimated rate of natural soil formation, about 1 mg ha^{-1} yr^{-1} (Troech and Thompson, 1993; Huang et al., 2015). It was not until 1999 that the European Conservation Agriculture Federation started promoting CA and adoption was visible in France, Finland and Spain where the growth of CA in perennial crops, such as fruit orchards, vineyards and olive plantations has exceeded the adoption rate in annual crops (Friedrich et al., 2012).

1.2. The rise and impacts of conservation agriculture

Conservation agriculture was introduced as a practice that aimed soil protection after the devastative dustbowl, it consists of any type of tillage system in which at least 30% of the soil surface is covered with crop residues after harvest (Frazee et al., 2005). These systems include no-till, strip-till, ridge tillage and mulch tillage or any other system designed to retain all or a portion of the previous crop's residue on the soil surface. The portion required depends on other conservation practices that may be included in the farmer's total conservation plan (Bradford and Peterson, 2000).
These tillage systems can prevent losses of arable land while regenerating degraded lands as they promote maintenance of a permanent soil cover, minimum soil disturbance, and diversification of plant species. They enhance biodiversity and natural biological processes above and below the ground surface, which contribute to increased water and nutrient use efficiency and to improved and sustained crop production (FAO, 2016).

By 2008/2009, the global extent of CA cropland covered about 106 M ha (Table 1.1) corresponding to 7.5% of global cropland (Kassam et al., 2015). During the last decade, this area witnessed an increase to reach 180.5 M ha (12.5% of global cropland) representing a difference of some 74 M ha over the 7-years period (Kassam et al., 2018). This worldwide expansion of CA is a glimpse of the advantages that this practice brings to both the environment and farmers as it improves the yearly input of new organic matter (OM), minimizes soil erosion and degradation, increases plant available water capacity, helps mitigate climate change, reduces cost of production and improves profitability and the stabilization of productivity on a long-term (Lal, 2007; Farooq and Siddique, 2015; Choudhary et al., 2017; Kassam et al., 2018).

Table 1.1. Global area distribution of conservation agriculture (M ha) by continent and year.

Continent	2008/2009	2013	2015/2016
South America	49.6	66.4	69.9
North America	40.0	54.0	63.2
Australia and New Zealand	12.1	17.9	22.7
Asia	2.6	10.3	13.9
Russia* and Ukraine	0.1	5.2	5.7
Europe	1.6	2.0	3.6
Africa	0.5	1.2	1.5
Total	106.5	157.0	180.5

* in 2008/2009, the area under CA in Russia was not available

Kassam et al., 2015; 2018

Introduction of CA practices in Europe was mainly driven by economic considerations. According to Soane and Ball (1998), reduced tillage and no-tillage practices as means of reducing crop production costs and allowing greater timeliness was intensively researched in many parts of Europe between 1960 and 1990. Scientific evidence of the long-term economic impacts of CA is rare at the European level (Tebrügge and Böhrnsen, 1997; Kächele et al. 2001; Nielsen et al. 2004a-b). But it seems clearly that except for Norway and Germany where reduced tillage is subsidised (Lundekvam et al. 2003; Schmidt et al., 2003), the reduction of production costs acts as a powerful driving force for CA adoption (Lahmar, 2010).

1.2.1. Influence of conservation agriculture on soil organic carbon

The global carbon (C) pool of 2500 Pg includes about 1550 Pg of soil organic carbon (SOC) and 950 Pg of soil inorganic carbon (SIC). The soil C pool is 3.3 times the size of the atmospheric pool (760 Pg) and 4.5 times of the biotic pool (Lal, 2004a). Severe depletion of SOC pool degrades soil quality, reduces biomass productivity, promotes water and wind erosion and adversely impacts water quality. The conversion of natural to agricultural ecosystems causes depletion of the SOC pool by as much as 60% in soils of temperate regions and 75% or more in cultivated soils of the tropic (Lal, 2004a). Both concentration and distribution of SOC has been found to be easily affected by tillage practices as it was reported that SOC recorded higher content in upper layers with no-tillage (NT) than with conventional tillage (CT), but a higher SOC content in the deeper layers of tilled plots where crop residues were incorporated through tillage (Jantalia et., 2007; Sombrero and de Benito, 2010).

Physical disturbance of soil structure through tillage results in a direct breakdown of soil aggregates and an increase of their turnover (Six et al., 2000a). Tillage brings subsurface soil to the surface where it is then exposed to wet-dry and freeze-thaw cycle and subjected to raindrop impact, and the absence of crop residues on the soil surface increases the susceptibility to further disruption (Paustian et al., 1997; Six et al., 2000a; Verhulst et al., 2010). Moreover, a redistribution of soil organic matter (SOM) takes place during tillage and these small changes can influence the stability of macro-aggregates and lead to a C loss and a gain of C-depleted micro-aggregates (Six et al., 2000b). The minimal or non-existent disturbance in CA reduces the destruction of soil aggregate structure, slows the turnover of macro-aggregates, prevents the decomposition of organic carbon (OC) by soil microorganisms and extends the storage period of OC in aggregates (Oades, 1984; Salinas-Garcia et al., 2002; Paterson, 2003). Therefore, the accumulation of SOM under NT compared to CT confers important improvements in soil quality, soil fertility and C sequestration (Six, 2000a).

One of the primary sources of OM inputs and precursors of the SOC pool is crop residue since their retention has been widely observed to maintain and/or increase SOC concentration. Blanco-Canquí and Lal (2007) assessed long-term (10 years) impact of three levels (0, 8 and 16 Mg ha^{-1} on a dry matter basis) of wheat straw applied annually on SOC under NT in central Ohio and reported SOC contents of 82.5 Mg ha^{-1} in the unmulched soil, 94.1 Mg ha^{-1} and 104.9 Mg ha^{-1} with 8 and 16 Mg ha^{-1} of crop residues respectively in the first 50 cm soil depth. Furthermore, Stetson et al. (2012) found that the removal of corn residue from the soil surface led to an increase of aggregate degradation, as the removal levels increased (from grain-only harvest to all aboveground corn biomass harvest) the potential for long-term soil productivity was negatively impacted. Thus, crop residue binds soil particles together into aggregates and protects SOC from mineralization (Stetson et al., 2012; Zhang et al., 2018). According to Lal (2009), returning crop residues to soil as amendment is essential to sequestering soil C at a rate of 100 - 1000 kg C ha^{-1} $year^{-1}$ depending on soil type and climate with a total potential of 0.6 - 1.2 Pg C $year^{-1}$ in world soils.

Based on the previous statements, retaining crop residue combined with low soil disturbance offers a way to increase soil C as some authors like Laird and Chang (2013) reported higher SOC content under NT with residue retention (58.4 g C kg^{-1}) than when incorporated in CT (54.3 g C kg^{-1}) while their removal resulted in the lowering of SOC in the 0-15 cm of soil in both tillage systems, Zhang et al. (2018) found that NT and ridge tillage with residue retention presented higher SOC content (48.2 and 48.3 Mg C ha^{-1}, respectively) in the first 20 cm soil depth than in mouldboard ploughed plots (45.7 Mg C ha^{-1}) in maize-soybean rotation system, Li et al. (2020) reported also that the combination of minimum tillage (NT and reduced tillage) and residue retention increased SOC stock by 13 and 12% respectively in the 0-30 cm soil, in comparison with CT in a meta-analysis that englobed 243 peer-reviewed publications covering all continents. However, these studies highlight the fact that conservation practices are more effective in the upper layers of the soil than in deeper ones which is the opposite of CT where SOC stocks are reported to increase in depth due to residue incorporation.

1.2.2. Influence of tillage practices on soil water content

Due to the continuous changes of climatic conditions manifested in an increase of global temperature, a decrease and an irregularity of precipitation, crop producers fear drastic drops of their productivity due to scarce water resources, which are one of the crucial factors that contributes to the decline in agricultural productivity. The current challenge in agriculture is to produce high yields by utilizing less water, especially in regions with limited land and water resources.

The necessity of insuring high or long-term stable crop productivity is also one of the reasons for the spread of CA, which not only contributes to decrease SOC loss and protect soil from wind erosion and degradation but can also increase infiltration and water retention on the soil surface and reduce runoff and evaporation compared to CT and NT with residue removal (Verhulst et al., 2010; Laird and Chang, 2013). Actually, when the soil surface is wet from a recent irrigation or precipitation event, evaporation from bare soil will occur at a rate controlled by atmospheric demand (Figure 1.2). The evaporation rate decreases as the soil surface dries over time while with residue retention, the soil surface is shielded from solar radiation and air movement just above the soil surface is reduced (van Donk et al., 2010) and thereby buffer surface soil temperatures and decrease water loss to evaporation (Laird and Chang, 2013).

Some researchers highlighted the importance of crop residue retention, Gicheru (1994) showed that maize stover used as a mulch resulted in more soil moisture down the profile (0-120 cm) throughout both short rain period (October-December) and long rain period (March-May) during two years than CT and tied ridges in a semi-arid area of Kenya. Mupangwa et al. (2007) studied the effect of both different rates of maize mulching and tillage on soil water content (SWC) in a clay and sandy soils in Zimbabwe, and reported that mulching helped conserve soil water in a season with long drought period at both experimental sites and that SWC consistently increased with increase in surface cover across the three tillage practices (planting basins, ripper tine and CT). Gava et al. (2013) found that cumulative values of evaporation decreased every time the density of soil coverage increased and went from 4 to 15%, 15 to 22% and 24 to 25% due to wheat straw retention in amounts of 2.5; 5 and 10 t ha^{-1}, respectively compared to bare soil.

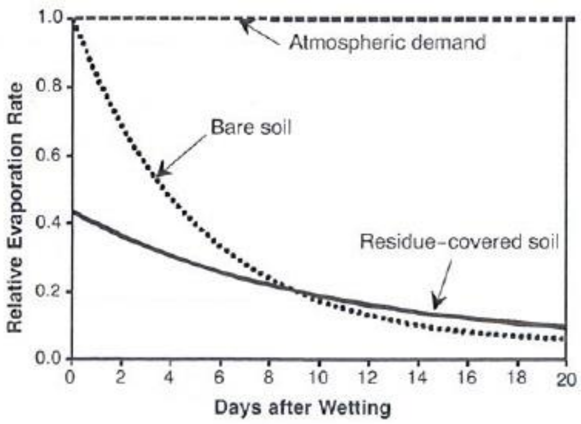

Figure 1. 2. Evaporation rates, relative to atmospheric demand, from bare soil and residue-covered soil after a single wetting event, a conceptual diagram (adapted from Watts and Klocke, 2004).

However, Tolk et al. (1999) found that soil water under a mulched surface was being used for crop growth and yield rather than for evaporation of soil water which is the main intention of almost crop producers, on this basis, researchers focused their studies on assessing the effect of conservation practices and crop retention on SWC thus crop productivity. De vita et al. (2007) carried out a three-year experiment on rainfed wheat crop planted in two different locations in southern Italy and found that NT with residue retained on the surface displayed greater SWC and wheat yield than CT ensuring also a good level of grain quality even when precipitation was <300mm. Van Donk et al. (2010) underlined the importance of residue retention combined with NT management as they reported higher maize yield and SWC in residue-covered plots than in bare soil plots in a two-year study. According to Johnson and Hoyt (1999) and Bradford and Petersen (2000) NT with residue retention decreases the frequency and intensity of short midseason droughts and may significantly affect crop yields during years of poor rainfall distribution.

1.3. Agriculture and climate change

“Warming of the climate system is unequivocal, and since the 1950s, many of the observed changes are unprecedented over decades to millennia. The atmosphere and ocean have warmed, the amounts of snow and ice have diminished, and sea level has risen”, this statement made by the IPCC (2014) describes the current situation the world is facing, as the global mean temperature has risen by 0.8ºC since the 1850s and is projected to increase by 1.5- 5.8ºC during the 21st century (Utomo, 2014) while the level of the build-up of heat trapping gases (greenhouse gases) in the Earth’s atmosphere continue to rise. Originally, greenhouse gases (GHGs) such as water vapor, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are naturally present in the Earth’s atmosphere and without them the average temperature of the Earth’s surface would be -18ºC rather than the present average of 15ºC (Seguin and Soussana, 2008), but in the last century humans’ activities have been interfering with the balance of these GHGs which give off additional emissions into the atmosphere.

By examining air bubbles in ice cores taken from Antarctica, scientists were able to estimate that atmospheric CO_2 concentration was never higher than 300 ppm (parts per million) throughout the 650,000 prior to industrialisation; multi-decadal to centennial-scale variations were less than 10 ppm (Solomon et al., 2007), however, since the pre-industrial era this concentration went from 280 ppm to 409.8 ppm in 2019 (Lindsey, 2020) while N2O concentration went from around 270 ppb (parts per billion) to 331 ppb in 2018 with the fastest growth observed in the last five decades (Tian et al., 2020). The human activities reinforced and accelerated the processes of global warming by the emission of long-lived GHGs such as CO_2, CH_4 and N_2O which are chemically stable and persist in the atmosphere for decades and centuries or longer, thus their emission has a long-term influence on climate, and short-lived gases like sulphur dioxide and carbon monoxide which are chemically reactive and generally removed by natural oxidation processes in the atmosphere, by removal at the surface or by washout in precipitation which make their concentrations highly variable (Solomon et al., 2007).

Anthropogenic GHG emissions are mainly generated from energy use in industry, transport and heat production and reached 73.2 % of global emissions in 2016 while agriculture, forestry and land use accounted for 18.4% (Figure 1.3). Although the agricultural sector is a secondary GHG emitter, it plays a fundamental part in the enhancement of climate change as it is both a producer and consumer of different forms of energy (direct and indirect) and with the increasing demand to feed a growing population, GHG emissions are likely to continue to rise from agroecosystems (Huang et al., 2018).

In this context, direct energy input is required to perform various operations related to tillage management and crop production such as seedbed preparation, irrigation, weed control, threshing, and harvesting and it includes mainly the use of fossil fuel, electricity and lubricants while indirect energy is used to produce farm inputs such as fertilizers, pesticides, seeds and machines (Singh, 2000; Khaledian et al., 2010; Mughal and Amjad, 2012). The use of both forms of energy (direct and indirect), microbial decomposition and the burning of stubble and OM in the soil (3.5% in 2016) are some of the emitters of CO_2 (Lal, 2004b), while most of N_2O is produced in soils through biological processes of nitrification and denitrification (4.1%in 2016) which are enhanced by the availability of nitrogen (N) from the application of synthetic N fertilizers, animal manure and the retention of crop residues (Signor and Cerri, 2013; Wang et al., 2013, Tian et al., 2020). Croplands, land use changes, fossil fuel combustion and fertilizers application produce most CO_2 and N_2O emissions (11.3% of the global GHG emissions from agriculture (Figure 1.3)), meanwhile intensive livestock and rice cultivation are the main sources for CH_4 emissions being 7.1% in 2016 (Climate Watch, 2020).

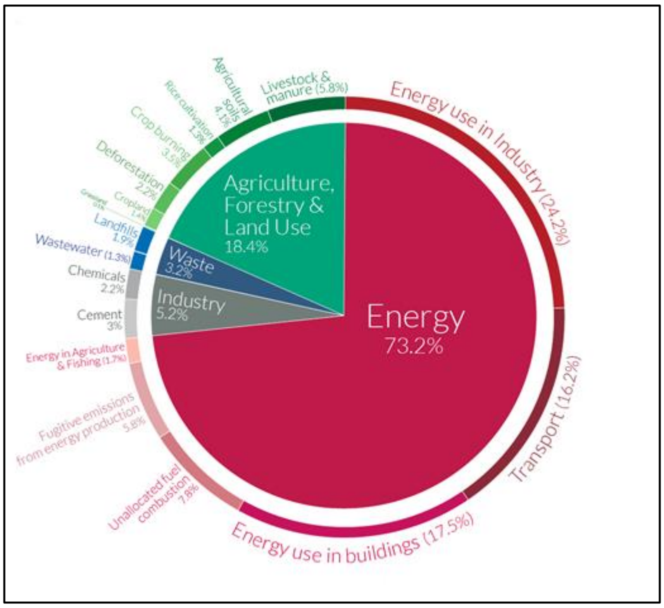

Figure 1.3. Global greenhouse gas emissions by sector in 2016 (Climate Watch, the World Resources Institute, 2020).

At the farm level, fuel use for tillage is responsible for most direct emissions and it depends on numerous factors such as soil properties, tractor size, type of the equipment used and depth of tillage (Lal, 2004b), moreover the fuel requirement increases with increase in depth of soil ploughing is for heavy than lighter textured soils (Collins et al., 1977). Many studies reported that fuel consumption is higher under CT where different equipments such as mouldboard plough, chisel and disk plough are employed than under conservation tillage where the use of machinery is reduced all along the crop cycle (Lobb, 1989; Köller, 1996; Rathke et al., 2007; Khaledian et al., 2010; Zhang et al., 2016). For instance, C emissions reached 35.3 kg CE ha^{-1} for a complete tillage including ploughing, two disking, field cultivation and rotary hoeing, 20.1 kg CE ha^{-1} by elimination of mouldboard ploughing and merely 5.8 kg CE ha^{-1} by elimination of mouldboard ploughing, disking, cultivation and hoeing, while seeding after chiselling would reduce the emissions from 35.3 to 7.9 kg CE ha^{-1} (Lal, 2004b).

In addition of CO_2 emissions, N_2O is emitted by soils as a result of the processes of denitrification in anaerobic soil and nitrification in aerobic soil with the anaerobic production considered more important (Ball et al., 1999). Most studies have shown that the larger N_2O emissions from NT compared to CT were associated to soil structure, application of N fertilizers and a greater SWC; Ball et al (1999) reported that N_2O emissions were high and accentuated by soil compaction and rainfall in a spring barley

(*Hordeum vulgare*) crop, Baggs et al. (2003) found that N_2O emissions were two to seven times higher from fertilized zero tillage treatments than from fertilized CT treatments in maize (*Zea mays*) plots and Escobar et al. (2010) also obtained higher N_2O emissions from a soybean (*Glycine max L. Merr.)* crop under NT system than under CT due to a high rate of denitrification and confirmed that the main driving variables of N_2O emissions are microbial biomass activity, soil temperature, water-filled pore space and soil nitrate content.

Greenhouse gas emissions are one of the key indicators in assessing the environmental sustainability of farming and can be evaluated by their carbon footprint (CFP) as an environmental performance index (Weinheimer et al., 2010; Gómez-Limón and Sanchez-Fernandez, 2010). However, according to Wiedmann and Minx (2008) there is no clear definition of the CFP as it requires a clear statement of underlying assumptions and often, the methodological approach. Still, some authors have agreed that the CFP should consider all emissions of a product both backward in time from the point of consumption to emission sources and forward in time to include the use and disposal phase of products (Peters, 2010; ISO, 2013), in other words the CFP expressed in CO_{2eq} of crop production could be quantified by considering the overall GHGs emissions from agricultural inputs used for crop production, protection and farm machinery in a single whole cycle of crop production (Adler et al., 2007).

The evaluation of the CFP of main crop products is commonly performed with a life cycle assessment (LCA) method (Figure 1.4) as it includes CO_2 emissions from off-farm manufacture, transportation and delivery of the different farm inputs as well as the emissions during the crop cultivation (Liu et al., 2016) that are derived from tillage operations, the use of synthetic fertilizers and phytosanitary products, crop residue decomposition, SOC gains or losses and from N_2O emissions nitrification, denitrification and mineralization (Figure 1.4).

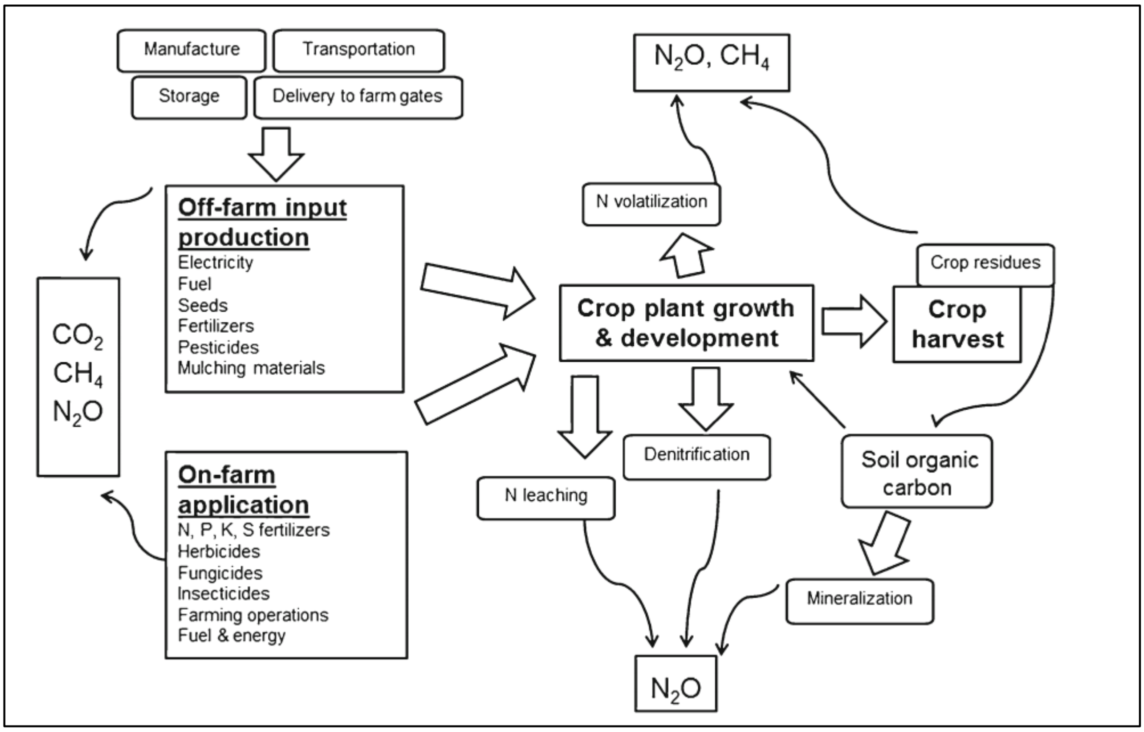

Figure 1.4. Main factors contributing to the carbon footprint of a crop production cycle (Liu et al., 2016).

As mentioned previously, tillage disturbance is the dominant factor reducing soil carbon stabilization within microaggregates, whereas conservation practices increase SOC content by enhancing C sequestration which plays a key role in reducing the CFP of crop cultivation because a per unit farmland GHG emission represents the balance between C-equivalent emissions and C sequestration in the production of a field crop (Liu et al., 2016).

The agricultural sector contributes to GHG emissions by (a) the combustion of fossil fuel and use of energy from alternate sources to assure the production, formulation, storage of farm inputs and the use of mechanized equipments (Lal, 2004b) and (2) tillage managements which have a major influence on soil C emissions as some authors reported higher CO_2 emissions under CT (Reicosky and Archer, 2007; Abdalla et al, 2013; Lu et al., 2015; Huang et al., 2018) than under NT system.

Intensive tillage breaks up the soil aggregates thus the SOM and exposes the fresh surface for enhanced gas change from the interior where the aggregates may contain higher CO_2 concentration (Reicosky and Archer, 2007). Moreover, La Scala et al., (2006, 2008) suggested that soil tillage accelerates OC oxidation, releasing large amounts of CO_2 to the atmosphere over few weeks thereby lowering the total C sequestration held within the soil. According to a review carried out by Holland (2004), significant reductions of CO_2 emissions were observed after the adoption of conservation tillage in different locations of Europe with various climate conditions and soil textures.

1.4. Maize and tillage practices

It is considered that maize (*Zea mays* L.) was one of the first plants cultivated by farmers between 7000 and 10,0 00 years ago, with evidence of maize as food coming from some archaeological sites in Mexico where some small corn cobs, estimated at more than 5000 years old, were found in caves (Ranum et al., 2014).

The spread of maize from Mexico to various parts of the world has been remarkable and rapid with respect to its evolution as a cultivated plant and as a variety of food products, such as cornflakes, corn syrup and oil for human consumption and for animal feed, it is highly desirable because of its high yield and feed value of grain, leaf and stem (Huang et al., 2006). Moreover, an important part of maize production is being used to generate ethanol fuel which is a biofuel additive for gasoline. A strong demand for ethanol production has resulted in increased maize prices and has provided incentives to increase maize acreage (Committee on World Food Security, 2013).
Nowadays, maize is one of the most three important cereal crop species after wheat and rice and is planted throughout a wide range of climates with the United States, China, Brazil and Argentina being the main producers as their production accounted for 793,949 thousand tons corresponding to over two thirds of global production in 2020 (World Data Atlas, 2020).

Usually, maize is cultivated in sites with available water whether from rainfall or irrigation, as it is considered more susceptible to water stress than other crops because of its unusual floral structure with separate male and female floral organs (Huang et al., 2006). Water stress lasting for many days can lead to smaller canopy cover during the vegetative stage and the reduction of the stomatal conductance which can reduce the crop transpiration and photosynthesis. This of course leads directly to reduced rate of biomass production, hence reduced grain yield (Hsiao, 2012). Although, over the last few decades maize grain yield has continued to increase thanks to a higher planting density, improved fertilization, optimal canopy structure and late-maturing cultivars with longer life cycles, some arid areas are still facing the irregularity of rainfall and suffering from water scarcity. This has driven some farmers to convert from CT to CA, considering that conservative practices have higher holding water capacity which is crucial to the crop in times of stress and in the most critical period that coincides with the interval of ten to fourteen days before and after flowering (Huang et al., 2006).

1.5. Conservation agriculture in Spain

A gradual change in the Spanish agriculture began in the 1950s, when prices rapidly increased, and the surplus labour pool began to shrink, as a half million rural field hands migrated to the cities or went abroad in search of a better life. Nonetheless, more substantial changes did not take place prior to the 1960s. The Stabilization Plan of 1959 encouraged emigration from rural areas, and the economic boom in both Spain and Western Europe provided increased opportunities for employment (Solsten and Meditz, 1988). The resulting lack of a ready labour supply was an incentive to mechanise, particularly for large landed estates as the number of farm tractors and harvester-threshers expanded more than tenfold between 1960 and 1983, from 52,000 to 593,000 and from 4,600 to 44,000, respectively. The process of mechanization caused agricultural productivity to grow by 3.5 percent per year between 1960 and 1978, and the productivity of farm workers grew even faster (Solsten and Meditz, 1988).

In 1960, within a context of incipient industrialisation and an economy that was relatively closed to other countries, oil accounted for just 29% of Spain's primary energy compared with 40% at a global level. However, between 1960 and 1973 there was a sharp rise in oil consumption as a result of unprecedented economic development. This sharp increase along with the increase of mechanization of agriculture consisted of a major concern for farmers as the cost were incrementing and there was an urge to lower them. As a response to this situation, conservation practices were established in many areas of Spain, especially in semi-arid regions where rainfall is another limiting factor for crop production, since they promote OM accumulation on the soil surface, reduce the deterioration of soil structure due to erosion and save time and energy which becomes an advantage from an economic point of view. Madejón et al. (2009) found that long-term adoption of conservation tillage (reduced tillage and NT) in Mediterranean areas of Spain has an effective way to increase SOM and, especially, to improve biochemical quality at the soil surface, Sombrero and de Benito (2010) also confirmed that conservation tillage is highly effective in enhancing SOC at a depth of 0-30 cm in a 10-year study in the semi-arid conditions of Castile and Leon and Giráldez et al, 1997 declared that NT has the ability to conserve water in low rainfall seasons in southern Spain and to protect the soil against erosion in wet seasons. Moreover, CA could increase crop yield in arid areas as reported by Fernández-Ugalde et al. (2009) who stated that barley yields under NT (2000 kg ha^{-1}) were sometimes twice those obtained under CT (1000 kg ha^{-1}) within areas of extreme aridity in northern Spain (the Ebro Valley).

Not only excessive tillage declines the soil structure and quality, but it also contributes to the GHGs emissions as López-Garrido et al. (2009) found that tillage caused a sharp increase in soil CO_2 emissions immediately after tillage in south Spain. Throughout the year, cumulative losses of carbon through CO_2 emissions were higher under CT than under NT and reduced tillage. Álvaro-Fuentes et al. (2004, 2007ab) also reported that soil CO_2 emissions just after tillage were 40% higher under CT than under NT because the CO_2 accumulated in soil pores was released to the atmosphere after the tillage event in a study of short- and long-term CO_2 fluxes in north east Spain. At the same time, tillage has an accumulative effect during the whole growing season in increasing microbial decomposition resulting in 20% higher soil CO_2 emissions under CT than under NT. Part of this effect can be attributed to greater root respiration under ploughing, especially in warmer months (Almaraz et al., 2009).
In addition of CO_2 emissions, farmers tend to apply high quantities of synthetic fertilizers which increase N availability in the soil, stimulate the nitrification and denitrification processes and result in greater N_2O emissions whether from leaching or runoff. In 2017, these emissions reached 12.420 CO_{2eq} (kt) and were 4.6% higher than the ones recorded in 2016 (MAPAMA, 2018). Although the amounts of N released from the soil are low, it has a high potential in global warming being 298 times more harmful than CO_2.

Irrigated agricultural systems have a major importance in Spain as the area under irrigation was 3.828.747 ha in 2019 corresponding to 22.5% of the total cultivated area (MAPAMA, 2019). However, the water availability is sometimes restricted during periods of high temperature and is only accessible for human consumption as it was the case during the summer of 2017 in Valladolid province of Castile and Leon, this was due to the low rainfall recorded in both winter and spring of the same year. The effect of this restriction was observed in the data collected by the *Ministerio de Agricultura, Pesca y Alimentación* of maize grain yield which reached 7.239 kg ha^{-1} in 2017 compared to 12.406 kg ha^{-1} in 2018. As a response to these unexpected decisions or even to the irregularity of rainfall in Castile and Leon some farmers have converted their fields to conservation practices as better conservation of water in the soil directly influences the increase of production in CA compared to conventional agriculture. In irrigated crops, a similar behaviour could be expected in conservation systems, thus an increase of the water use efficiency and a reduction of economic and energy expenses and for that, the monitoring and quantification of soil water plays a fundamental role in the agricultural research.

To summarize all the cited above, conservation agriculture could offer the possibility of restoring SOC by sequestering carbon in the soil at a rate ranging from 0.1-0.5 t ha^{-1} $year^{-1}$ (Kassam et al., 2018) and keeps the rate of soil loss under that of soil formation (Kertész et al., 2011) when compared to conventional agriculture, thus enhancing the long-term sustainability of the system. It also plays an important part in buffering the emissions of GHGs related to excessive tillage and reducing the carbon footprint of a crop production cycle and enhances the biodiversity and natural biological processes above and below ground. Furthermore, soil water content is almost always greater at planting in NT which increase moisture conservation resulting in greater crop yields (Bradford and Petersen, 2000). Higher input factor productivities with low levels of inputs in CA can provide a greater return to investment and a more robust basis for sustainable production intensification (Kassam et al., 2014).

Conservation agriculture can achieve the improvements listed above only when the three principles cited by the FAO are present and which are:

- Minimum mechanical soil disturbance: Minimum soil disturbance refers to low disturbance no-tillage and direct seeding. The disturbed area must be less than 15 cm wide or less than 25% of the cropped area (whichever is lower). There should be no periodic tillage that disturbs a greater area than the aforementioned limits. Strip tillage is allowed if the disturbed area is less than the set limits.
- Permanent soil organic cover: Three categories are distinguished: 30-60%, >60-90% and >90% ground cover, measured immediately after the direct seeding operation. Area with less than 30% cover is not considered as CA
- Diversification of crop species: implemented by adopting a cropping system with crops in rotation.

The scarcity of the studies conducted on the effect of tillage systems on the physicochemical soil properties, nitrate leaching, water dynamics, CO_2 emissions and carbon footprint in a continuous irrigated maize crop in Castile and Leon, was the main reason why this study was implemented.

References

Abdalla, M., Osborne, B., Lanigan, G., Forristal, D., Williams, M., Smith, P., Jones, M.B. 2013. Conservation tillage systems: a review of its consequences for greenhouse gas emissions. Soil Use and Management, 29: 199-209.

Adler, R., Del Grosso, S.J., Parton, W.J. 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecological Applications, 17: 675-691.

Almaraz, J.J., Zhou, X., Mabood, F., Madramootoo, C., Rochette, P., Ma, B-L., Smith, D.L., 2009. Greenhouse gas fluxes associated with soybean production under two tillage systems in southwestern Quebec. Soil & Tillage Research, 104: 134-139.

Álvaro-Fuentes, J., López, M.V., Gracia, R., Arrúe, J.L., 2004. Effect of tillage on short-term CO2 emissions from a loam soil in semi-arid Aragón (NE Spain), Options Méditerranéennes, Serie A, 60, 51-54.

Álvaro-Fuentes, J., Cantero-Martínez, C., López, M.V., Arrúe, J.L., 2007a. Soil carbon dioxide fluxes following tillage in semiarid Mediterranean agroecosystems. Soil & Tillage Research, 96: 331-341.

Álvaro-Fuentes, J., López, M.V., Arrúe, J.L., Cantero-Martínez, C., 2007b. Management effects on soil carbon dioxide fluxes under semiarid Mediterranean conditions, Soil Science Society of America Journal, 72: 194-200.

Baggs, E.M., Stevenson, M., Pihlatie, M., Regar, A., Cook, H., Cadisch, G. 2003. Nitrous oxide emissions following application of residues and fertilizer under zero and conventional tillage. Plant and Soil, 254: 361-370.

Ball, B.C., Scott, S., Parker, J.P. 1999. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. Soil & Tillage Research, 53: 29-39.

Blanco-Canqui, H., Lal, R. 2007. Impacts of long-term wheat straw management on soil hydraulic properties under no-tillage. Soil Science Society of America Journal, 71: 1166-1173.

Bradford, J. M., Peterson, G. A. 2000. Conservation tillage. In Handbook of soil science, ed. M. E. Sumner, G247-G269. Boca Raton, FL, USA: CRC Pres

Carter, M. R., Rennie, D. A. 1982. Changes in soil quality under zero tillage farming systems: distribution of microbial biomass and mineralizable C and N potentials. Canadian Journal of Soil Science, 62: 587-597.

Climate Watch, 2020. Historical GHG emissions: https://www.climatewatchdata.org/ghg-emissions?breakBy=sector&end_year=2016&start_year=1990)

Collins, N.E., Kemble, L.J., Williams, T.H. 1976. Energy requirements for tillage on coastal plains soils. *In:* Lockeretz W, editor. Agriculture and Energy. New York: Academic Press; 1976. p. 233-244.

Committee on World Food Security. High Level Panel of Experts (HLPE). 2013. Biofuels and food security. A report by the High-Level Panel of Experts on food security and nutrition of the Committee on World Food Security. Rome. Accessed December 17, 2013. http://www.fao.org/fileadmin/user_upload/hlpe/hlpe_documents/HLPE_Reports/HLPE-Report-5_Biofuels_and_food_security.pdf

Derpsch, R. 2004. History of crop production, with and without tillage. Leading Edge, 3: 150-154.

De Vita, P., Di Paolo, E., Fecondo, G. Di Fonzo, N., Pisante, M. 2007. No-tillage and conventional tillage effects on durum wheat yield, grain quality and soil moisture content in southern Italy. Soil & Tillage Research. 92, 69-78

Duley, F. L., Fenster C. R. 1954. Stubble Mulch Wheat Farming Methods for Fallow Areas. Historical Materials from University of Nebraska-Lincoln Extension Service and USDA. 35 pp. Revised Version E. C. 54-100.

Escobar, L.F., Carneiro Amado, T.J., Bayer, C., Chavez, L.F., Acordi Zanatta, J., Ernani Fiorin, J. 2010. Postharvest nitrous oxide emissions from a subtropical Oxisol as influenced by summer crop residues and their management. Revista Brasileira de Ciência do Solo, 34: 435-443.

FAO. 2016. What is Conservation Agriculture? http://www.fao.org/conservation agriculture/overview/what-is-conservation-agriculture/en/

Fernández-Ugalde, O., Virto, I., Imaz, M.J. Enrique, A., Bescansa, P. 2009. Soil water retention characteristics and the effect on barley production under no-tillage in semi-arid conditions of the Ebro Valley (Spain). Proceedings of 18th ISTRO Conf., Izmir, Turkey, Paper T1-008.

Frazee, R., Towery, D., Peterson, D., Hirschi, M. 2005. U.S. Tillage Trends. Land and Water. University of Illinois Extension, University of Illinois, Bull. Number 6.

Fowler, P. 2002. Farming in the first millennium AD. Cambridge, UK: Cambridge University Press.

Friedrich, T., Derpsch, R., Kassam, A. 2012. Overview of the Global Spread of Conservation Agriculture. Field Actions Science Reports. The journal of filed actions. Special Issue 6. 2012.

Gava, R., De Freitas, P.S.L., De Faria, R.T., Rezende, R., Frizozone, J. A. 2013. Soil water evaporation under densities of coverage with vegetable residue. Engenharia Egrícola Jaboticabal v.33, n.1, p.89-98.

Gicheru, P.T. 1994. Effects of residue mulch and tillage on soil moisture conservation. Soil Technology, Volume 7, Issue 3: 209-220.

Giráldez, J.V., González, P. Ordóñez, R., Laguna, A., de Haro, J.M., 1997. Conservation tillage under extreme meteorological conditions in Southern Spain. *In:* Tebrügge, F., Böhrnsen, A. (Eds.), Experiences with the Applicability of No-tillage Crop Production in the West-European Countries. Proceedings of the EC-Workshop III, Wissenschaftlicher Fachverlag, Giessen, Germany, pp. 119-125.

Gómez-Limón, J.A., Sanchez-Fernandez, G. 2010 Empirical evaluation of agricultural sustainability using composite indicators. Ecological Economics 69 (5): 1062-1075.

Harari, Y. V. 2015. Sapiens: A Brief History of Humankind. Harper Collins Publ. USA; Illustrated edition. 464 pp.

Holland, M. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. Agriculture, Ecosystems & Environment, 103(1): 1-25.

Hsiao, T.C. 2012. Maize In: Steduto, P., Hsiao, T.C., Fereres, E., Raes, D. (Eds.), Crop yield response to water. Food and Agriculture Organization of the United Nations, Rome, pp. 114-120.

Huang, M., Liang, T., Wang, L., Zhou, C. 2002. Effects of no-tillage system on soil physical properties and carbon sequestration under long term wheat maize double cropping system. Catena, 128: 195-202.

Huang, R., Birch, C.J., George D.L. 2006. Water use efficiency in maize production – The challenge and improvement strategies. *In:* Water to Gold- Proceedings 6th triennial Conference, Maize Association of Australia

Huang, Y., Ren, W., Wang, L., Hui, D., Grove, J.H., Yang, X., Tao, B., Goff, B. 2018. Greenhouse gas emissions and crop yield in no-tillage systems: A meta-analysis. Agriculture, Ecosystems and Environment, 268: 144-153.

IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151.

ISO TS 14067. 2013. Greenhouse Gases-carbon Footprint of Products-requirements and Guidelines for Quantification and Communication. International Organization for Standardization, Geneva, Switzerland.

Jantalia, C. P., Resck, D. V. S., Alves, B. J. R., Zotarelli, L., Urquiaga, S., Boddey, R. M. 2007. Tillage effect on C stocks of a clayey Oxisol under a soybean-based crop rotation in the Brazilian Cerrado region. Soil & Tillage Research, 95: 97-109.

Johnson, A.M., Hoyt, G.D. 1999. Changes to the soil environment under conservation tillage. HortTechnology, 9: 380-393.

Kächele, H., Frielinghaus, M., Kühn, G., 2001. Economic assessment of soil conservation systems in Northeast Germany. Proceedings, Intern. Symposium ESSC Multidisciplinary Approaches to Soil Conservation Strategies. Helming ed. ZALF-Report 47, 115-120.

Khaledian, M. R., Mailhol, J. C., Ruelle, P., Mubarak, I., Perret S. 2010. The impacts of direct seeding into mulch on the energy balance of crop production system in the SE of France. Soil & Tillage Research. 106(2), 218–226.

Kassam, A., Derpsch, R., Friedrich, T. 2014. Global achievements in soil and water conservation: The case of Conservation Agriculture. International Soil and Water Conservation Research, 2(1): 5-13.

Kassam, A., Friedrich, T., Derpsch, R., Kienzle, J. 2015. Overview of the worldwide Spread of Conservation Agriculture. Field Actions Science Reports. The journal of field actions. Vol8. 2015.

Kassam, A., Friedrich, T., Derpsch., R. 2018. Global spread of Conservation Agriculture. International Journal of Environmental Studies, Volume 76: 1-23.

Kertész, Á., Madarász, B., Bádonyi, K., Csepinszky, B., Mika, J. 2011. Conservation tillage for rational water management and soil conservation. Hungarian Geographical Bulletin, 60(2): 117-133.

Köller, K. 1996. Production des céréales sans labour. Revue Suisse d'Agriculture, 28 : 30-34.

Laird, D. A., Chang, C. W. 2013. Long-term impacts of residue harvesting on soil quality. Soil & Tillage Research, 134, 33–40.

Lahmar, R. 2010. Adoption of conservation agriculture in Europe. Lessons of the KASSA project. Land Use Policy, 27:4 -10.

Lal, R. 1974. No-Tillage effects on soil properties and maize (*Zea mays L.*) production in Western Nigeria. Plant Soil 40: 321-331.

Lal, R. 2004a. Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. Soils - The Final Frontier Viewpoint. Science Vol 304, 1623-1627.

Lal R. 2004b. Carbon emission from farm operations. Environment International, 30(7), 981-990.

Lal, R., Reicosky, D., Hanson, J. 2007. Evolution pf the plow over 10,000 years and the rationale for no-till farming. Soil & Tillage Research. 93:1-12.

La Scala, N., Bolonhezi, D., Pereira, G.T. 2006. Short-term soil CO2 emission after conventional and reduced tillage of a no-till sugar cane area in Southern Brazil. Soil & Tillage Research, 91: 244-248.

La Scala, N., Lopes, A., Spokas, K., Bolonhezi, D., Archer, D.W., Reicosky, D.C. 2008. Short-term temporal changes of soil carbon losses after tillage described by a first-order decay model. Soil & Tillage Research, 99: 108-118.

Lerche, G. 1994. Ploughing Implements and Tillage Practices in Denmark from the Viking Period to About 1800: Experimentally Substantiated. Commission for Research on the History of Agricultural Implements and Field Structures, vol.8. Royal Danish Academy of Sciences and Letters: Poul Kristensen Herning.

Li, Y., Li, Z., Chang S.X., Cui, S., Jagadamma, S., Zhang, Q., Cai, Y. 2020. Residue retention promotes soil carbon accumulation in minimum tillage system : Implications for conservation agriculture. Science of the Total Environnent, 740 : 1-11.

Lindsey, R. 2020. Climate Change: Atmospheric Carbon Dioxide. NOAA. Climate.gov. Science and information for a climate-smart nation. https://www.climate.gov/news-features/understanding-climate/climate-change-atmosphericcarbondioxide#:~:text=Before%20the%20Industrial%20Revolution%20started,the%20Industrial%20Revolution%20in%201750

Liu, C., Cutforth, H., Chai, Q., Gan, Y. 2016. Farming tactics to reduce the carbon footprint of crop cultivation in semiarid areas. A review. Agronomy for Sustainable Development 36:69.

Lobb, D. 1989. A Study of the Impact of No-till on Tractor Fuel Cost vs. Crop Returns. Agricultural Energy Centre, Guelph, Ontario, Canada. 19 pp.

López-Garrido, R., Díaz-Espejo, A., Madejón, E., Murillo, J.M., Moreno, F., 2009. Carbon losses by tillage under semi-arid Mediterranean rainfed agriculture (SW Spain). Spanish Journal of Agricultural Research, 7: 706-716.

Lu, X., Lu, X., Khan Tanveer, S., Wen, X., Liao, Y. 2015. Effects of tillage management on soil CO_2 emission and wheat yield under rain-fed conditions. Soil Research, 54(1): 38-48.

Lundekvam, H.E., Romstad, E., Oygarden, L. 2003. Agricultural policies in Norway and effects on soil erosion. Environ. Sci. Policy 6, 57-67.

Madejón, E., Murillo, J.M., Moreno, F., López, M.V., Arrúe, J.L., Álvaro-Fuentes, J., Cantero Martínez, C. 2009. Effect of long-term conservation tillage on soil biochemical properties in Mediterranean Spanish areas. Soil Tillage Research, 105: 55-62.

Mannering, J. V. 1979. Conservation Tillage to Maintain Soil Productivity and Improve Water Quality. Historical Documents of the Purdue University Cooperative Extension Service. Department of Agricultural Communication. 7 pp.

MAPAMA, 2018. Informe anual de indicadores: https://www.mapa.gob.es/es/ministerio/servicios/analisis-y-prospectiva/informe_anual_indicadores2018__tcm30-513683.pdf

MAPAMA, 2019. Encuesta sobre superficies y rendimientos de cultivos. Informe sobre regadios en España.

Montgomery D.2007. *Dirt: The Erosion of Civilizations.* Berkeley and Los Angeles, California: University of California Press.

Mupangwa, W., Twomlow, S., SueWalker, S., Hove, L. 2007. Effect of minimum tillage and mulching on maize (*Zea mays L.*) yield and water content of clayey and sandy soils. Physics and Chemistry of the Earth, Parts A/B/C. Volume 32, Issues 15-18: 1127-1134.

Nielsen, V., Mortensen, H. and Sørensen, K., 2004a. Fuel consumption, time consumption and capacity by using reduced tillage and direct sowing. DJF report, Landbrug. 105, 89 pp.

Nielsen, V., Mortensen, H. and Sørensen, K., 2004b. Reduced tillage: fuel consumption and time consumption. Grøn Viden, Markbrug. 294, 8 pp.

Oades, J. M. 1984. Soil organic matter and structural stability: mechanisms and implications for management. Plant and Soil, 76: 319-337.

Paterson, E. 2003. Importance of rhizodeposition in the coupling of plant and microbial productivity. European Journal of Soil Science, 54:741-750.

Paustian, K., Collins H. P., Paul, E. A. 1997. Management controls on soil carbon. P. 15-49. *In* E. A. Paul et al. (ed.) Soil organic matter in temperate agroecosystems: Long-term experiments in North America. CRC Press, Boca Raton, Fl.

Peters, G. P. 2010. Carbon footprints and embodied carbon at multiple scales. Current Opinion in Environmental Sustainability, 2: 245-250.

Ranum, R., Peña-Rosas, J.P., Garcia-Casal, M.N. 2014. Global maize production, utilization, and consumption. Annals of the New York Academy of Sciences, p: 105-112.

Rathke, G.W., Wienhold, B.J., Wilhelm, W.W., Diepenbrock, W.2007. Tillage and rotation effect on corn–soybean energy balances in eastern Nebraska. Soil & Tillage Research, 97(1): 60-70.

Reicosky, D.C., Archer, D.W. 2007. Moldboard plow tillage depth and short-term carbon dioxide release. Soil & Tillage Research, 94(1): 109-121.

Salinas-Garcia, J. R., Velazquez-Garcia, J. D., Gallardo-Valdez, A., Diaz-Mederos, P., Caballero-Hernández, F., Tapia-Vargas, L. M., Rosales-Robles, E. 2002. Tillage effects on microbial biomass and nutrient distribution in soil under rain-fed corn production in central-western Mexico. Soil & Tillage Research, 66(2) : 143-152.

Schmidt, W., Nitzsche, O., Zimmerling, B., Krück, S., 2003. Implementation of conservation tillage as erosion control strategy on cropland in Saxony. Saxon State Agency for Agriculture, Leipzig, Germany. 9pp.

Seguin, B., Soussana, J.F. 2008. Emissions de gaz à effet de serre et changement climatique : causes et conséquences observées pour l’agriculture et l’élevage. Courrier de l’environnement de l’INRA 55 : 79-91

Signor, D., Pellegrino, Cerri C. E. 2013. Nitrous oxide in agricultural soils: a review. Pesquisa Agropecuária Tropical Goiânia v. 43, 3, 322-338.

Singh, J.M. 2000. On farm energy use pattern in different cropping systems in Haryana, India. [PhD Thesis] Germany, International Institute of Management, University of Flensburg.

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

Six, J., Paustian, K., Elliott, E. T., Combrink, C. 2000b. Soil structure and Organic Matter: I Distribution of Aggregate-Size Classes and Aggregate-Associated Carbon. Soil Science Society of America Journal, 64: 681-689.

Soane, B.D., Ball, B.C., 1998. Review of management and conduct of long-term tillage studies with special reference to a 25-yr experiment on barley in Scotland. Soil & Tillage Research, 45: 17-37.

Solomon, S., Qin, D., Manning, M., Alley., R.B., Berntsen, T., Bindoff, N.L., Chen, Z., Chidthaisong, A., Gregory, J.M., Hegerl, G.C., Heimann, M., Hewitson, B., Hoskins, B.J., Joos, F., Jouzel, J., Kattsov, V., Lohmann, U., Matsuno, T., Molina, M., Nicholls… Wratt, D. 2007: Technical Summary. *In:* Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the FourthAssessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Solsten, E., Meditz, S.W. 1988. Spain: A Country Study. *In:* Solsten, E and Meditz, S. W..editors. Spain: A Country Study. Washington: GPO for the Library of Congress, p 159-160.

Sombrero, A., de Benito, A. 2010. Carbon accumulation in soil. Ten-year study of conservation tillage and crop rotation in a semi-arid area of Castile-Leon, Spain. Soil & Tillage Research, 107: 64-70.

Stetson, S. J., Osborne, S.L., Schumacher, T.E., Eynard, A., Chilom, G., Rice, J., Nichols, K.A., Pikul, J.L. Corn residue removal impact on topsoil organic carbon in corn-soybean rotation. Soil Science Society of America Journal 76 (4), 1399-1406.

Tebrügge, F., Böhrnsen, A., 1997b. Crop yields and economic aspects of no-tillage compared to plough tillage: Results of long-term field experiments in Germany. Proceedings of the EC workshop II. Experience with the applicability of no-tillage crop production in the West-European countries. Silsoe, UK. Tebrügge and Böhrnsen eds. Wissenschaftl. Fachverlag Giessen, 25-44.

Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., Davidson, E.A., Ciais, P., Jackson R.B., Janssens-Maenhout, G., Prather, M.J., Regnier, P., Pan, N., Pan, S., Peters, G.P., Shi, H., Tubiello, F.N., Zaehle, S., … Yao, Y. 2020. A comprehensive quantification of global nitrous oxide sources and sinks. Nature, 586: 248-272.

Tolk, J. A., Howell, T.A., Evett, S.R. 1999. Effect of mulch, irrigation, and soil type on water use and yield of maize. Soil & Tillage Research, 50: 137-147.

Triplett Jr., G. B., Van Doren Jr., D. M. 1969. Nitrogen, Phosphorus and Potassium fertilization of non-tilled maize. Agronomy Journal. 61, 637-639.

Troech, F.R., Thompson, L.M. 1993. Soil and Soil Fertility. New York: Oxford University Press. 489pp.

van Donk, S.J., Martin, D.L., Irmak, S., Melvin, S.R., Petersen, J.L, Davison, D.R. 2010. Crop residue cover effects on evaporation, soil water content and yield of deficit-irrigation corn in west-central Nebraska. American Society of Agricultural and Biological Engineers. Vol.53 (6): 1787-1797.

Verhulst, N., Govaerts, B., Verachtert, E., Castellanos-Navarrete, A., Mezzalama, M., Wall, P., Deckers, J., Sayre, K.D., 2010. Conservation Agriculture, Improving Soil Quality for Sustainable Production Systems? In: Lal, R., Stewart, B.A. (Eds.), Advances in Soil Science: Food Security and Soil Quality. CRC Press, Boca Raton, FL, USA, pp. 137-208.

Utomo, M. 2014. Conservation Tillage Assessment for Mitigation Greenhouse Gas Emission in Rainfed Agro-Ecosystems. *In:* Kaneko N., Yoshiura S., Kobayashi M. (eds) Sustainable Living with Environmental Risks. Springer, Tokyo.

Wang, Y.Y., Hu, C.S., Ming, H., Zhang, Y.M., Li, X.X., Dong, W.X., Oenema, O. 2013. Concentration profiles of CH_4, CO_2 and N_2O in soils of a wheat–maize rotation ecosystem in North China Plain, measured weekly over a whole year. Agriculture, Ecosystems & Environment, 164: 260-272.

Watts, D., Klocke, N.L. 2004. Chpter6: Crop water use *In* Irrigation Management Home Study course. C. Burr, ed. Publication EC98-788-S. Lincoln, Neb.: University of Nebraska Cooperative Extension.

Weinheimer, J., Rajan, N., Johnson, P., Maas, S. 2010. Carbon footprint: a new farm management consideration in the Southern High Plains. Agricultural & Applied Economics Association 25-27.

Wiedmann, T. Minx, J. 2008. A Definition of 'Carbon Footprint'. In: C. C. Pertsova, Ecological Economics Research Trends: Chapter 1, pp. 1-11, Nova Science Publishers, Hauppauge NY, USA.

World Data Atlas, 2020. https://knoema.com/atlas/topics/Agriculture/Crops-Production-Quantity-tonnes/Maize-production

Zhang, X. Q., Pu, C., Zhao, X., Xue, J.F., Zhang, R., Nie, Z.J., Chen, F., Lal, R., Zhang, H.L. 2016. Tillage effects on carbon footprint and ecosystem services of climate regulation in a winter wheat-summer maize cropping system of the North China Plain. Ecological Indicators, 67: 821-829.

Zhang, Y., Li, X., Gregorich, E. G., McLaughlin, N.B., Zhang, X., Guo, Y., Liang, A., Fan, R., Sun, B. 2018. No-tillage with continuous maize cropping enhances soil aggregation and carbon storage in Northeast China. Geoderma, 330: 204-2011.

SCOPE OF THE THESIS

1. Objectives

The main objective of this Thesis is to *study the effect of management strategies of conservation agriculture compared to conventional agriculture in irrigated maize monoculture on the soil quality, water dynamics, CO_2 emissions and the carbon footprint in semi-arid conditions of Castile and Leon.*

Different research works have been carries out with the purpose of achieving the main objective:

i. Evaluating the effect of conventional tillage (CT) and no-tillage (NT) practices on the soil properties and nitrate leaching.

ii. Studying the influence of CT and NT managements on the dynamics of soil water content and maize grain yield.

iii. Evaluating the effects of CT and NT managements on the soil organic carbon changes, CO_2 emissions and its relation with both soil temperature and moisture.

iv. Estimating the total greenhouse gas emissions produced from the agricultural energy inputs, the N_2O emissions and SOC changes in the soil in relation with grain yield expressed in $tCO_{2eq}\ t^{-1}$. The estimation of these components of the CFP is determined under CT and NT using different N fertilization rates.

2. Thesis outline

The present thesis has been divided into different chapters in order to study the different aspects of a maize production cycle under both CT and NT systems and to carry out the objectives listed above**.** All chapters, except I, II and VIII, have the same structure consisting of six sections (introduction, material and methods, results, discussion, conclusion and references).

In ***Chapter 1***, a general overview of the evolution of agriculture and the gradual rise of conservation agriculture and the advantages that it is bringing to the agricultural and environmental sectors is presented. In Scope of the thesis the main objectives and outline of the thesis are presented.

Chapter2 is dedicated to a general description of the experiment site as well as the study of the effect of CT and NT on the different soil properties and quality and nitrate leaching.

In ***Chapter 3*** the effect of tillage system on the SWC and water balance at different soil depths as well as the maize yield components are studied.

Chapter 4 aims to analyse the response of CO_2 emissions and its relation with both soil temperature and moisture under CT and NT managements as well as the soil organic carbon accumulation. It should be mentioned that this chapter include data that were collected prior to the start date of the thesis (since 2012) as it was interesting to see the evolution of CO_2 emissions on a long-term experiment.

The effect of tillage systems and different rates of synthetic fertilization on the carbon footprint of the irrigated maize monoculture is assessed in ***Chapter 5,*** and the same as the previous chapter, this one also chapter include data that were collected prior to the start date of the thesis (since 2012).

Finally, general conclusions are presented in ***Chapter 6***.

CHAPTER 2. EFFECT OF TILLAGE SYSTEM ON SOIL PROPERTIES AND NITRATE LEACHING IN A CONTINUOUS IRRIGATED MAIZE CROP

2.1. Introduction

In 1938, a soil scientist and chief of the USDA's (United States Department of Agriculture) bureau for chemistry and soil by the name of Kellogg stated: *"Essentially, all life depends upon the soil… There can be no life without soil and no soil without life; they have evolved together",* this statement has never been more accurate as nowadays since the soil is a critical part for a successful agriculture that is feeding more than 7.7 billion persons over the world (United Nations, 2021). Furthermore, the rising demand for agricultural products led to extensive clear-felling of forests and conversion into arable land and meadows which caused multiple changes that can manifest through the reduction of soil chemical and physical properties, leading to soil quality decline and continuing reduction of productivity (Tolimir et al. 2020). The soil loss can be considered as an irreversible process because the loss of 1 mm of topsoil can take 3000 years to replace, hence, the soil is considered a very slow to almost non-renewable resource (Baxter and Willamson, 2001).

Soil-forming processes require time, which does not necessarily refer to a specific period, such as months or years, but a degree of soil weathering which consists of chipping away rock fragments and thus modify its inherent physical and chemical characteristics. These processes take hundreds or even thousands of years to happen (Kalev and Toor, 2018). In addition of the minerals originated from the weathering of remains of parent rocks, soil is also formed of the organic matter (OM) which consists of the remains of the living organisms in various stages of decomposition as well as living micro-organisms. From a physical point of view, OM improves the aeration of soils, increases the water holding capacity and contributes to aggregates stability by supplying food for microorganisms (NRCS, 2006).

Human activities affect soil formation in numerous ways. Some of the anthropogenic soil alteration practices include the destruction of natural vegetation, tillage, mixing, and compaction (Kalev and Toor, 2018). Lester Brown (as cited by Kaiser, 2004) estimated that human activity was responsible for the loss of 26 billion tons of topsoil per year, 2.6 times the natural rate. The frequent cultivation of croplands drove to soil losses by water and wind erosion as the vegetation is often removed before and after every crop cycle which leave a bare soil susceptible to degradation by climatic factors besides to human overexploitation (Pimentel and Burgess, 2013). The degradation of soil structure and the negative nutrient balance, deplete soil fertility, exhaust soil organic carbon (SOC) pool and reduce soil and crop productivity (Lal, 2009) and despite the availability of improved varieties with increased yield potential, the potential increase in production is generally not attained because of poor crop management (Reynolds and Tuberosa, 2008). However, nowadays, some farmers concerned about the environmental sustainability of their crop production systems combined with the ever-increasing production costs have begun to adopt and adapt improved system management practices such as conservation agriculture which led to ultimate vision of sustainable agriculture (Verhulst et al., 2010).

Conservation agriculture is a widely used terminology to denote soil management systems that result at least 30% of the soil surface being covered with crop residues after seeding the subsequent crop (Jarecki and Lal, 2003). And to achieve this level of ground cover, conservation tillage normally involves some degree of tillage reduction and the use of non-inversion tillage methods (Verhulst et al., 2010). According to Lal (2009), crop residue retention and no-tillage (NT) influence crop yield by moderating micro-climate and soil processes, they have short-term impact on crop yields and long-term impact on agronomic productivity and sustainability. Short-term impacts on crop yields vary among seasons depending on soil temperature and moisture regimes, rainfall amount and distribution and incidence of pests and diseases while in the long-term, residue retention improves agronomic productivity and enhances sustainability because of improvements in soil quality. No-tillage and residue retention ameliorate soil physical, chemical and biological quality, improves use efficiency of inputs and decrease their costs, increase productivity per unit area, time and energy use and preserve soil carbon (C) (Lal, 2009; Liu et al., 2014) while conventional tillage (CT) causes physical breakdown of the soil structure which makes it susceptible to erosion due to dis-integration of soil aggregates (Bronick and Lal, 2005). Although CT results in better structural distribution than reduced tillage and NT, the components of the soil structure in CT are very weak to resist water slacking resulting in structural deterioration (Six et al., 2000; Verhulst et al., 2010).

Soil quality has been defined by Karlen et al. (1997) as the capacity of a specific kind of soil to function within natural managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation. Within the framework of agricultural production, high soil quality equates to

the ability of the soil to maintain high productivity without significant soil or environmental degradation (Verhulst et al., 2010). The assessment of soil quality is based on the physical, chemical and biological characteristics of the soil and which are considered as soil quality indicators. According to Doran and Parkin (1994), these indicators should include ecosystem processes such as C and nitrogen (N) cycling, be sensitive to climatic changes and soil management and be accessible and understandable to many users and applicable in field conditions. Doran and Parkin (1996) developed a list of basic soil properties that meet many requirements of indicators for screening soil quality and health and it includes physical indicators such as soil texture, bulk density (BD) and water holding capacity, chemical indicators which encompass soil organic matter (SOM), total organic C and N, electrical conductivity, pH, and extractable N, phosphorus (P) and potassium (K), while the biological indicators consist of the microbial biomass C and N, soil respiration and potentially mineralizable N.
The soil quality indicators aforementioned were found to be influenced by tillage practices as Rhoton (2000) reported a 10% loss of initial SOM content with plough tillage during the first four years of tillage. Some authors reported that soil Bd would increase under NT system (Dam et al., 2005; Gozubuyuk et al., 2014; Salem et al., 2015) while others like Huang et al. (2015) reported a soil BDdecrease after nine years of NT in silty loam soil in China.

The excessive exploitation of the soil may not only lead to its deterioration but can also consist of an inconvenient for human health, because a poor soil structure is prone to high nitrate leaching (NO_3^-). It is true that more than half the world is nourished by crops grown with synthetic N fertilizers; unfortunately, unintended adverse environmental and human health impacts result from the escape of reactive N from agricultural soils, including groundwater contamination, eutrophication of freshwater and estuarine ecosystems, tropospheric pollution related to emissions of nitrogen oxides and ammonia gas, and accumulation of nitrous oxide (Griffis et al., 2013; Steffen et al., 2015)

In this context, it was interesting to assess the evolution of soil properties and nitrate leaching under both CT and NT systems in a continuous irrigated maize crop in semiarid conditions of Castile and Leon. Therefore, the main objective of this chapter is to study the effect of the tillage practices cited above in addition of different N-fertilization rates on the soil quality and nitrate leaching.

2.2. Materials and methods

2.2.1. Experimental design and crop management

This study, initiated in 2011, was carried out during 2015-2017, in the experimental field of the Agricultural Technological Institute of Castile and Leon (41º42’23’N, 4º41’36’W) in the Spanish province of Valladolid. The experiment was set up on a Typic Xerofluvent soil (USDA 87 classification). The experimental design included blocks randomly chosen (Figure 2.1) of 16 plots of 144 m^2 of continuous irrigated maize and where the main factor of the study was tillage system (CT and NT).

Figure 2. 1. Aerial view of the experiment location in 2016 (google earth Pro).

At the fall of every year, the seedbed of CT treatment was prepared with mouldboard plough, which involved complete soil inversion and burial of the previous maize residues up to a 30 cm depth, followed by the passing of cultivator that stirred and pulverized the soil in the following spring and before planting. The second treatment consisted of NT in which weeds were chemically controlled (glyphosate application (2.5 l ha^{-1})) in the fallow period (from 4 to 10 days before planting) and the crop was directly planted on the standing residues of the previous one. It is necessary to mention that in November 2010, a vetch crop (*Vicia sativa*) was sowed before sowing maize in April 2011 in order to homogenize the experimental field. Under CT management this crop was incorporated into the soil using a mouldboard plough, while under NT management, the crop residues remained on the soil surface after the application of glyphosate.

Every year, a distance of 55 cm between maize rows and 22 cm between plants were left giving an average plant population of 90.000 plants ha^{-1}. The setting of the crop was achieved using a "Nodet" conventional drill in CT plots and a "Semeato" no-till seed drill in NT plots. The variety Roxxy was planted in 2015 and 2016 while in 2017 it was changed to the variety Lexxtour. Moreover, the set 8-15-15 of N, P_2O_5 and K_2O was applied in every plot at a rate of 800 kg ha^{-1}and herbicides such as Camix (4% Mesotrione, 40 % S-metolachlor), Closar (Chlorpyrifos 48%), Emblem (Bromoxynil 20%) and Karate Zeon (10% p/v Lambda Cyhalothrin) were applied to prevent from the development of some maize diseases. Sprinkler irrigation was established according to the crop hydric needs and the meteorological conditions, and the amount of water applied was at a rate of 4.5 l h^{-1}.

The application of top-dressing N fertilization occurring during the month of June, included a conventional dose (FC and FE) used by the crop producers of the region and a reduced amount (FR and FER) consisting of 700 (FC) and 600 (FR) kg ha-1 of Calcium Ammonium Nitrate N27 (NAC 27%) from 2012 to 2014. From 2015 to 2017, the synthetic fertilization consisted in the application of 700 kg ha^{-1} Calcium Ammonium Nitrate N27 (NAC 27%) (FC), which is characterized by its versatility of use that includes almost every soil type and different crops that require the availability of an immediate uptake of N (50% nitric) and later slower uptake (50% ammoniacal), 700 (FE) and 600 (FER) kg ha^{-1} of Ammonium Sulfate Nitrate (ENTEC 26%). Unlike the NAC 27%, the ENTEC fertilizer contains the DMPP molecule (3, 4- dimethylpyrazole phosphate) which inhibit the Nitrosomonas soil bacteria responsible of the transformation of the ammoniacal N into nitric oxide. The N stabilized in ENTEC remains in the soil up to several months under the ammoniacal form delaying its transformation into nitrate.

In the first year of the trial, 2015, the maize was planted on the 6th of May and harvested on the 25th of November. The irrigation treatment started on the 3rd week of May and ended in the last week of September, the crop received a total amount of 749.2 mm of water from irrigation. The second year, the sowing happened on the 28th of April and harvest on the 9th of November 2016, a total of 682 mm of water was applied starting on the 2nd week of June until the 3rd week of September. In the last year, the planting took place on the 3rd of May and the harvest on the 10th of October 2017; a total 384.7 mm of water was brought to the crop from the 1st week of May until the 2nd week of August. In 2017, the irrigation was cut-off during the grains' filling, due to the decision of the organization responsible of the regulation of water resources (*Confederación*

Hidrografica del Duero) of shutting down the water designated to irrigation because of the drought recorded during the summertime.

At the outset of the study and during the following years, the different soil chemical properties were determined by collecting soil samples up to 100 cm soil depth after the maize harvest in 2011, 2015 and 2017 at three sites in each elementary plot to obtain a composite sample per plot. The samples were air dried and sieved through a 2 mm mesh, then went through different analytical processes in order to obtain the SOM, total C, total N, available phosphorus (P) and extractable potassium (K).

To determine the yield components, plants samples were picked in one-meter area from four rows. Afterwards, plant, ear numbers, rows per ear and grain numbers per row were counted. Furthermore, in every plot, two strips of 12m x 1.5 m were harvested and kernels were weighed separately to estimate the crop yield.

2.2.2. Soil texture determination

The soil texture which describes the relative proportions of sand, silt and clay was determined by the size and type of particles that make up the soil. During the testing procedure, sand particles which diameter is comprised between 2.0-0.05 mm are retained on a sieve while silt (0.5-0.002 mm diameter) and clay (<0.002 mm diameter) particles pass through. Silt and clay are separated via particle settlement. Afterwards, the percentages of these particles were determined using the rate of sedimentation based on Stoke's law which describes the relationship between the frictional force of a sphere moving in a liquid and other quantities (such as particle radius and velocity of the particle).

Once, the percentages of sand, silt and clay obtained, the textural triangle suggested by the FAO (Figure 2.2) was used to classify the soil texture.

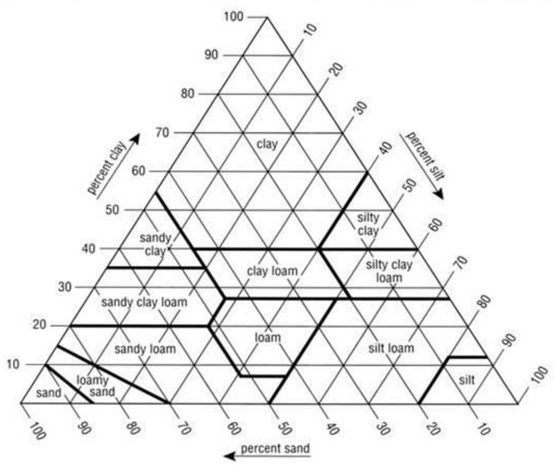

Figure 2.2: Guide to soil texture determination

2.2.3. Soil bulk density

Soil bulk density (Bd) was determined in 2015 and 2017 using the core method (Blake, 1965) under both tillage systems, undisturbed core samples were taken carefully from the soil profile at six depths levels (0-10, 10-20, 20-30, 30-60 and 60-100 cm). The core sampling was made using cylinders of 5.35 cm diameter and 3 cm length as shown in Figure 2.3. The core samples were immediately weighed, and then dried at 105 °C for 24 h to a constant weight and were reweighed. Afterwards, soil Bd was calculated according to the following equation:

$$Bd = \frac{M}{V} \quad (1)$$

where Bd is soil bulk density (g cm^{-3}), M is the mass of the dry soil sample (g) and V is the volume of the sample (cm^3).

In 2015 the core samples were collected in January during the fallow period while in 2017, they were collected in July under a standing maize crop.

Figure 2.3. Soil sampling for soil bulk density determination in 2017.

2.2.4. Chemical properties

- *Soil pH*

From the soil samples that have been collected, soil pH was measured in a 1:2.5 (w/v) soil to 0.01mol/L CaCl2 suspension with a glass electrode.

- *Soil organic matter*

In 2011, the determination of the soil organic carbon was based on the Walkley and Black (1934) chromic acid wet oxidation. Oxidizable organic carbon (OC) in the soil was oxidised by 0.167 M potassium dichromate (K_2Cr2O7) solution in concentrated sulfuric acid. The heat of the reaction raises the temperature which is sufficient to induce substantial oxidation. The Cr2O72- reduced during the reaction with soil is proportional to the oxidizable organic C present in the sample. The organic carbon can then be estimated by measuring the remaining unreduced dichromate by back-titrating with ferrous sulphate or ammonium ferrous sulphate using diphenylamine or o-phenanthroline-ferrous complex as an indicator.

Since 2013, the total C content was determined by dry combustion using a LECO CNS 1934 analyzer. The determination of the OM content in the soil was achieved by multiplying the OC by 1.724.

- *Total nitrogen*

Samples were weighed into a porcelain sample holder (boat) for introduction into the combustion chamber utilizing an automated sample loader (CNS 2000, Operation Manual, Leco, St. Joseph, MI, USA). The sample is burned at high temperature (between

900 and 1000 °C or 1400 and 1600 °C) in an atmosphere of pure oxygen. Under these conditions, all C-containing compounds are completely decomposed and converted into carbon oxides (mainly carbon dioxide). The autoanalyzer measures and reports the total organic carbon value based on the concentration of carbon oxides present using various procedures (for example, a C gas detector and thermal differences between gas columns).

The combustion process converts covalently bound nitrogen into nitrogen gas (N_2). The N_2 is quantitated by passing the gas through a conductivity cell. Both carbon and nitrogen contents were expressed in g kg^{-1}.

- *Nutrient contents: phosphorus and potassium*

The content of available phosphorus (P) was determined by Olsen method (Olsen et al., 1982) which consists of the extraction of phosphate from the soil by 0.5 N sodium bicarbonate solution adjusted to pH 8.5. In the process of extraction, hydroxide and bicarbonate competitively desorb phosphate from soil particles and secondary absorption is minimized because of high pH.
Potassium (K) is extracted from air-dried soil samples with 0.5M ammonium acetate/acetic acid solution. This effectively displaces the potentially available K^+ ions. The potassium content of the filtered extract is then determined using a flame photometer. Both P and K contents were expressed in mg kg^{-1}.

- *Equivalent soil mass determination*

Ideally, management-induced changes in soil elements can be assessed from comparisons among similar soil (i.e., identical original thickness, BD, texture) with contrasting management histories. Thus, management effects can be deduced simply from changes in element concentration in the surface horizons when the soil masses are considered identical (if the changes in horizons thickness compensate for changes in BD); however, soil management alters the genetic horizons and the masses of the surface horizons (Ellert and Bettany, 1995) thus conduct to wrong comparison between soil managements.

To account for different soil masses, the amounts of OM, N, P and K (t ha^{-1}) were calculated in an equivalent mass of soil under both tillage systems according to the following equation:

$$M_{element} = [conc] \cdot Bd \cdot T \cdot 10000\ m^2\ ha^{-1} \cdot 1000\ kg\ t^{-1} \qquad (2)$$

where $M_{element}$ is the element mass per unit area (t ha^{-1}), [conc] is the element concentration (g kg^{-1}), Bd is the bulk density (t m^3) and T is the thickness of the soil layer (m).

The mass of the heaviest soil layer which was most susceptible to the influence of management was designated as the "equivalent" mass (Ellert and Bettany, 1995) and the additional soil thickness required to attain this equivalent mass in lighter soil layers was calculated as follow:

$$T_{add} = \frac{(M_{soil,equiv} - M_{soil,surf}) \cdot 0.0001\ ha\ m^{-2}}{BD_{subsurface}} \quad (3)$$

where T_{add} is the additional thickness of subsurface layer required to attain the equivalent soil mass (m), $M_{soil,\ equiv}$ (equivalent soil mass) is the mass of the heaviest horizon (t ha^{-1}), $M_{soil,\ surf}$ is the sum of soil mass in surface layer(s) (t ha^{-1}) and $Bd_{subsurface}$ is the bulk density of subsurface layer (t m^{-3}).

2.2.5. Nitrate leaching measurement

To determine the groundwater contamination by nitrates, PVC pipes with a nominal diameter of 5.08 cm were installed, at a depth at 55, 105 and 155 cm soil depth considering that at this depth the water is not used by the crop and goes directly to deeper depths (drainage). The nitrate dosage in the water was carried out by creating a pressure of -0.6 bar in the tubes, after irrigation, all the water contained in each tube was extracted into an individual container. Once the samples were collected, they were taken to the laboratory for analysis of the different compounds.

2.2.6. Statistical analysis

Comparison of all the data collected from the soil analysis of both CT and NT for all the studied years were performed by analysis of variance (ANOVA) in Infostat, statistical software. Treatment means were separated using the Tukey test at the 5% significant level (*$P \leq 0.05$*).

2.3. Results

The statistical analysis didn't show significant effect of the N fertilization on the different aspects studied in this experiment, and for that it wasn't included in the results and discussion sections.

2.3.1. Soil texture and bulk density

2.3.1.1. Texture

The soil texture presented in Table 2.1 did not show significant differences among neither the studied years nor tillage systems. However, the initial percentages of silt and clay displayed significant differences among soil depths under both tillage systems in 2011 while in 2015, the clay percentage was the only parameter to show significant difference under NT system. In 2011, the silt percentage was significantly higher in the upper 40 cm of the soil profile under CT and NT systems while the clay percentage was significantly higher in the lower layers of the soil profile under CT (at 100 cm) and NT (at 60 and 100 cm). Although the existence of the statistical differences, the initial soil texture was mostly silty loam in all the soil layers under CT and loam under NT. In 2015, the soil particles percentages witnessed a slight numerical change, nevertheless, the soil texture was not affected and remained as the described at the start of the experiment.

Table 2.1. Soil texture according at different depths of the soil profile in 2011 and 2011 under conventional tillage (CT) and no tillage (NT).

Tillage System	Depth (cm)	2011			2015		
		Silt (%)	Sand (%)	Clay (%)	Silt (%)	Sand (%)	Clay (%)
CT	10	51.33	32.00 a	18.67 c	46.00	34.00	20.33
	20	50.00	32.33 a	19.50 c	46.33	33.33	20.33
	30	50.83	31.83 a	19.83 c	47.33	32.50	20.50
	40	49.17	30.33 a	20.67 bc	47.33	31.33	21.50
	60	51.83	26.67 b	23.17 b	46.83	28.67	24.33
	100	51.67	19.00 b	28.00 a	48.67	28.67	21.17
NT	10	49.67	29.67 a	18.83 b	51.83	31.17	17.00 bc
	20	47.83	30.00 a	20.00 b	50.50	31.33	18.00 bc
	30	48.17	29.83 a	19.67 b	50.50	30.17	19.33 bc
	40	49.33	30.50 a	20.33 b	49.50	29.83	20.33 ab
	60	50.17	24.00 b	24.33 a	49.83	26.50	23.83 a
	100	49.17	23.50 b	25.00 a	49.33	28.67	21.17 ab

Different letters within the same column indicate statistical differences among depths ($P \leq 0.05$).

2.3.1.2. Soil bulk density

At the beginning of the experiment, the soil bulk density did not display significant difference between tillage systems neither between the different soil layers as shown in Table 2.2. As in 2011, the Bd recorded in 2015 was not affected neither by tillage systems nor by soil depths. In 2017, the pattern changed as the mean Bd was 5.88% higher under

CT than the one recorded under NT system at 0-100 cm soil depth and was also 10.96% higher than the reported in 2011 in the tilled plots at the same depth. Depending on the measurements obtained at the different soil depths, the mean Bd was the highest at 10-20 cm soil depth under CT and NT systems and the lowest at 60-100 cm soil depth under both tillage systems while there were no significant differences reported all along the rest of the soil layers under both tillage systems (Table 2.2).

Table 2.2. Soil bulk density in the soil profile under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)					
		Bulk density (g cm^{-3})					
		0-10	10-20	20-30	30-40	40-60	60-100
2011	CT	1.27	1.47	1.41	1.57	1.52	1.54
	NT	1.24	1.39	1.55	1.52	1.54	1.55
2015	CT	1.50	1.70	1.57	1.49	1.49	1.56
	NT	1.59	1.67	1.61	1.53	1.47	1.55
2017	CT	1.64 ab	1.70 a	1.69 ab	1.62 ab	1.55 ab	1.52 b
	NT	1.58 ab	1.60 a	1.54 ab	1.55 ab	1.48 ab	1.42 b
Tillage system			CT			NT	
Depth (cm)			0-100			0-100	
Mean	2011		1.46 B			1.52 B	
	2015		1.55 A			1.57 A	
	2017		1.62 A			1.53 B	

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among tillage system ($P \leq 0.05$).

2.3.2. Soil chemical properties

2.3.2.1. The soil pH

At the beginning of the experiment, the soil pH fluctuated between 8.22 and 8.63 under CT and between 8.43 and 8.68 under NT system as shown in Table 2.3. Although, no significant difference was reported, the pH was numerically higher in the non-tilled plots than in the tilled ones. In addition, the depth of sampling presented significant difference as the pH observed in the deeper layers (60 - 100 cm) was higher than in the upper layers especially under CT.

In 2015, the soil pH was 4.19 and 3.7 % higher under CT and NT systems respectively than in 2011, with the non-tilled plots presenting a higher pH mean (Table 2.3) than the tilled ones. Moreover, it can be observed that the pH displayed significantly elevated value at 60-100 cm soil depth than at 0-10, 10-20, 20-30, and 30-40 cm under CT, while it kept the same pattern as 2011 under NT system.

In 2017, the soil pH was measured up to the first 30 cm soil layer due to a high dryness that prevented from getting to further depths, no significant differences were observed between CT and NT systems neither between soil depths; however, the soil pH recorded in 2017 was 3.1 and 4.2% lower than in 2015 under CT and NT systems, respectively and 2% lower than the pH reported in 2011 under NT system (Table 2.3).

Table 2.3. Soil pH in the studied soil profile under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)					
		pH					
		0-10	10-20	20-30	30-40	40-60	60-100
2011	CT	8.24 b	8.22 b	8.24 b	8.26 b	8.42 ab	8.63 a
	NT	8.43	8.44	8.45	8.45	8.55	8.68
2015	CT	8.55 b	8.56 b	8.53 b	8.57 b	8.90 ab	9.03 a
	NT	8.67	8.72	8.75	8.82	8.93	9.02
2017	CT	8.27	8.29	8.29	-	-	-
	NT	8.30	8.33	8.39	-	-	-
Tillage system		CT			NT		
Depth (cm)		0-30	0-100		0-30	0-100	
Mean	2011	8.24 B	8.34 B		8.44 B	8.50 B	
	2015	8.55 A	8.69 A		8.71 A	8.82 A	
	2017	8.28 B	-		8.34 C	-	

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among years within a column ($P \leq 0.05$).

2.3.2.2. The soil organic matter and total nitrogen

In 2011, the mean SOM recorded under NT system at 0-60 cm soil depth was 15.5% higher than under CT as shown in Table 2.4 and where it can be observed that at 0-10 cm soil depth, the mean SOM was 30.5% higher in the non-tilled plots than in the tilled ones, however, at 40-60 cm soil depth the mean SOM was significantly lower than the recorded in the upper soil layers under NT system while it didn’t show significant differences under CT although the SOM value was numerically the lowest at 40-60 cm soil depth.

In 2015, the SOM content recorded under CT was 12.8% higher than the obtained in 2011 while no significant difference was observed under NT system for both years at 0-60 cm soil depth (Table 2.4). In this year, the tillage system did not affect the SOM content. However, it can be observed that the SOM was significantly the highest in the upper soil layers especially up to 30 cm soil depth compared to deeper ones (40-60 cm) under both tillage systems.

In the last year of the experiment, the mean SOM content was 35.58 and 37.37 % higher than the recorded in 2011 and 2015 respectively, under NT system at 0-30 cm soil depth while it was 35.87 % higher in 2017 than in 2011 under CT system (Table 2.4). Besides being the highest under NT system, the SOM content was 24.06% higher in the non-tilled plots than in the tilled ones. Considering soil depths, mean SOM accumulation was the most elevated under NT system especially in the first 10 cm of the soil profile while no significant difference was observed between soil depth under CT system.

Table 2.4. Mean equivalent mass of the soil organic matter (t ha^{-1}) under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)				
		Soil organic matter				
		0-10	10-20	20-30	30-40	40-60
2011	CT	15.00 B	15.96 B	15.30 A	15.90 A	13.83 A
	NT	19.58 Aa	19.32 Aa	17.89 Aa	18.14 Aa	12.86 Ab
2015	CT	18.01 a	19.72 a	19.71 a	17.10 a	11.29 b
	NT	20.04 a	19.28 ab	18.20 ab	15.71 bc	11.97 c
2017	CT	21.00 B	21.14 B	20.76 A	-	-
	NT	29.23 Aa	26.19 Aab	22.58 Ab	-	-
Tillage system		CT			NT	
Depth (cm)		0-30	0-60		0-30	0-60
Mean	2011	46.27 B’	75.99 B’		56.78 B	87.78 A
	2015	57.46 A’	85.78 A’		57.53 B	85.23 A
	2017	62.87 A’	-		78.00 A	-

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among tillage system within the same year depth ($P \leq 0.05$).
Different uppercases’ indicate statistical difference among years within tillage system ($P \leq 0.05$).

In 2011, the mean equivalent mass of the total nitrogen (TN) was 17.5 % higher under NT system than CT system in the first 10 cm of the soil profile while no significant difference between tillage systems was observed for the other soil layers as shown in Table 2.5. It can be observed that under CT system, the mean TN was the most elevated in the deeper layer of the soil profile (40-60 cm) while the lowest was recorded in the upper layers except at 10-20 cm soil depth where no significant difference where observed. As for NT system, the mean TN was also the highest at 40-60 cm soil depth and the lowest at 10-20 and 30-40 cm soil depth.

The mean TN equivalent mass in 2015, did not display significant difference neither between tillage systems nor between soil depths; nevertheless, it was 32.50 and 30.40% lower under CT and NT systems, respectively compared to 2011 at 0-60 cm soil depth (Table 2.5).

In the last year of the experiment, the mean TN equivalent mass was 28.8 % higher under NT system than CT system in the first 10 cm of the soil profile and no significant difference was noticed compared to the previous years at 0-30 cm soil depth. In this year, the TN was significantly the highest in the first 10 cm of the soil profile under NT system while no significant difference was observed between soil depths under CT system as presented in Table 2.5.

Table 2.5. Mean equivalent mass of the total nitrogen (t ha^{-1}) under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)				
		Total nitrogen				
		0-10	10-20	20-30	30-40	40-60
2011	CT	1.20 Bb	1.58 Aab	1.27 Ab	1.17 Ab	1.99 Aa
	NT	1.41 Aab	1.32 Ab	1.61 Aab	1.22 Ab	1.87 Aa
2015	CT	1.16	1.09	1.10	1.04	1.19
	NT	1.30	1.07	1.18	0.96	1.19
2017	CT	1.18 B	1.21 A	1.14 A	-	-
	NT	1.52 Aa	1.26 Ab	1.08 Ab	-	-
Tillage system		CT			NT	
Depth (cm)		0- 30	0-60		0-30	0-60
Mean	2011	4.05	7.21 A’		4.33 A’	7.42 A’
	2015	3.36	5.60 B’		3.54 B’	5.69 B’
	2017	3.52	-		3.85 AB’	-

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among tillage system within the same year depth ($P \leq 0.05$).
Different uppercases’ indicate statistical difference among years within tillage system ($P \leq 0.05$).

2.3.2.3. Nutrient contents : Available phosphorus and extractable potassium

In 2011, the mean equivalent mass of the available P was affected by tillage system and was 48.97% higher in the tilled plots than in the non-tilled ones as given in Table 2.6 at 0-60 cm soil depth. The highest amounts of this component were situated in the first 10 cm of the soil profile but witnessed a decrease as the soil depth increased.

In 2015, the available P was 67.95% higher under CT than under NT system and was 43.2 and 27.1% higher under CT and NT systems, respectively than the reported P in 2011 at

0-60 cm soil depth. The mean equivalent mass of P did not vary between soil depths in CT system but it presented the highest value at 40-60 cm soil depth and the lowest at 30-40 cm in NT system (Table 2.6).

The soil P content had the same pattern in 2017 as the previous year as it was 64.61% higher under CT than the one recorded under NT system. Considering soil depths, the available P did not vary all along the 30 cm of the soil profile in the tilled plots while it was significantly the highest in the first 10 cm of the soil profile in the non-tilled plots as shown in Table 2.6.

Table 2.6. Mean equivalent mass of the available phosphorus (kg ha^{-1}) under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)				
		Available P				
		0-10	10-20	20-30	30-40	40-60
2011	CT	45.84 Aa	39.90 Aa	36.09 Aa	32.17 Aab	29.15 Ab
	NT	33.31 Ba	24.94 Bb	24.93 Ab	18.73 Bb	21.03 Ab
2015	CT	62.66 A	56.17 A	52.09 A	45.01 A	46.42 A
	NT	33.10 Bab	30.51 Bbc	29.33 Bbc	25.11 Bc	38.18 Ba
2017	CT	60.32 A	56.66 A	54.83 A	-	-
	NT	39.96 Ba	33.87 Bab	30.57 Bb	-	-
Tillage system		CT			NT	
Depth (cm)		0-30	0-60		0-30	0-60
Mean	2011	121.83	183.13 B’		83.17 B’	122.93 B’
	2015	170.95	262.39 A’		92.93 AB’	156.23 A’
	2017	171.82	-		104.38 A’	-

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among tillage system within the same year depth ($P \leq 0.05$).
Different uppercases’ indicate statistical difference among years within tillage system ($P \leq 0.05$).

At the beginning of the experiment, tillage system did not affect the mean equivalent mass of the extractable K. However, the K contained in the first 10 cm of the soil profile and at 40-60 cm soil depth presented the highest values compared to the recorded in the middle layers of the soil especially at 30-40 cm soil depth under both tillage systems as observed in Table 2.7.

In 2015, the K content was 19.6% higher under CT than under NT system and was 19.1% significantly higher than the recorded in 2011 for CT system at 0- 60 cm soil depth (Table 2.7). As in 2011, the K content was significantly higher in the first 10 cm and at 40-60 cm soil depth than at 30-40 cm depth under both tillage systems.

In the last year of the experiment, no significant difference was noticed between tillage systems and as in the previous years, the mean equivalent mass of the extractable K was the highest at 0-10 cm soil depth under NT system while no significant difference was observed between soil depth under CT system. The amount of extractable K reported in 2017 was 22.5% higher than in 2011 under CT system and 15.8 and 21.8% higher than the recorded in 2015 and 2011, respectively under NT system at 0-30 cm soil depth (Table 2.7).

Table 2.7. Mean equivalent mass of the extractable potassium (kg ha^{-1}) under conventional tillage (CT) and no tillage (NT) in 2011, 2015 and 2017.

Year	Tillage system	Depth in the soil profile (cm)				
		Extractable K				
		0-10	10-20	20-30	30-40	40-60
2011	CT	416.83 a	337.98 b	305.68 bc	272.44 c	437.26 a
	NT	448.66 a	310.30 b	299.84 b	249.73 c	422.26 a
2015	CT	527.94 Aa	391.63 Abc	387.82 Abc	339.30 Ac	461.99 Aab
	NT	475.99 Aa	349.86 Ab	287.67 Bbc	230.67 Bc	419.40 Ba
2017	CT	453.49	438.78	408.42	-	-
	NT	500.74 a	416.41 b	372.58 b	-	-
Tillage system		CT			NT	
Depth (cm)		0-30	0-60		0-30	0-60
Mean	2011	1060.50 B’	1770.21 B’		1058.80 B’	1730.79 A’
	2015	1307.40 A’	2108.68 A’		1113.51 B’	1763.54 A’
	2017	1300.67 A’	-		1289.72 A’	-

Different lowercases indicate statistical differences among depths within a row ($P \leq 0.05$).
Different uppercases indicate statistical difference among tillage system within the same year depth ($P \leq 0.05$).
Different uppercases’ indicate statistical difference among years within tillage system ($P \leq 0.05$).

2.3.3. Nitrate losses by leaching under CT and NT systems

The concentration of nitrate leaching was determined in the drained water at 55, 105 and 155 cm soil depth under CT and NT systems in 2015, 2016 and 2017 during the months of irrigation is presented in Figure 2.4. It can be observed that among the years of the study, 2015 recorded the highest NO_3^- concentration, especially under CT where the highest concentration reached 589.8 mg l^{-1} in July at 105 cm soil depth while it was 69.6 mg l^{-1} under NT system at the same depth and month. The same pattern was observed also in august where the NO_3^- concentration reached 399.4 and 34.8 mg l^{-1} under CT and NT systems, respectively at 155 cm soil depth.

In the second year of the experiment, the mean NO_3^- concentration didn't show significant difference between tillage systems but was 3.27 and 2.09 times lower than in 2015 under CT and NT systems, respectively. Nevertheless, NO_3^- concentration displayed high levels under CT at 55 cm soil depth (768.8 8 mg l^{-1}) while under NT no samples were collected at this depth due to the absence of water in the tube during the month June. In this year, some samples were missing because of the absence of water that could have been drained in the installed tubes during June, August and September at 105 cm depth under CT system.

The last year was characterized by the least and the most irregular amount and frequency of the irrigation applied due to the drought period during the summer of 2017, most of the samples were collected during June and July while in august a and as it can be observed in Figure 2.4, the mean NO_3^- concentration was 5.12 and 1.56 times lower than the recorded one in 2015 and 2016, respectively under CT while it was 1.21 and 2.54 times higher in 2017 than in 2015 and 2016, respectively under NT system.

In June the NO_3^- concentration was higher in the first 55 cm of the soil profile under CT system (137.4 mg l^{-1}) than NT system (121.7 mg l^{-1}) while at 105 and 155 cm soil depth, the it was higher under in the non-tilled soil reaching 120.6 and 44.8 mg l^{-1}, respectively. In July, the NO_3^- concentration displayed higher values under NT system at 55 and 155 cm soil depth being 171.5 and 56.0 mg l^{-1}, respectively compared to CT system which reached values of 82.0 and 53.4 mg l^{-1} at the same soil depths respectively; however, at 105 cm soil depth, the NO_3^- concentration was higher in the CT system at 105 cm with 109.2 mg l^{-1} compared to 56 mg l^{-1} in the NT system.

Because of the irregularity of the application of irrigation and its scarcity in August of 2017, some samples were not collected which explain the absence of data regarding the NO_3^- concentration in this month.

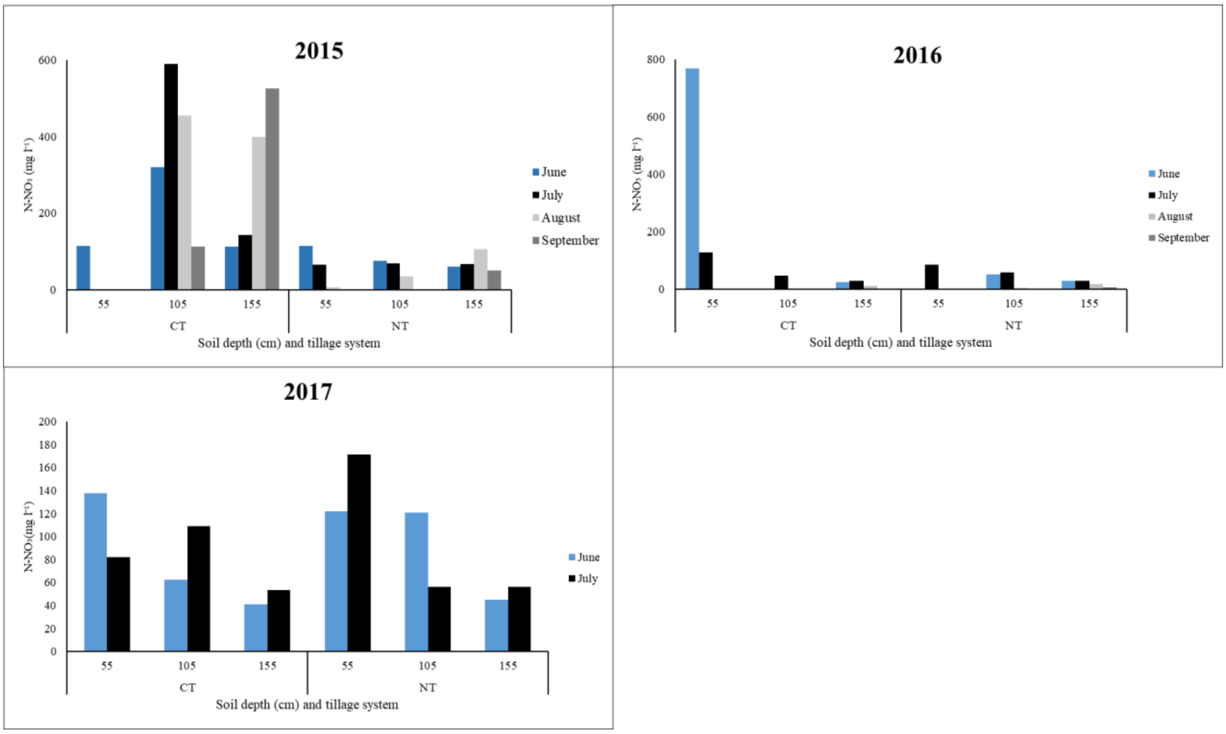

Figure 2.4. Mean nitrate (N-NO_3) concentration under conventional tillage (CT) and no-tillage (NT) at three soil depths in 2015, 2016 and 2017 during the irrigation months. Different letters indicate statistical differences among years depths ($P \leq 0.05$).

2.4. Discussion

2.4.1. Soil bulk density

As observed in Table 2.2, soil Bd didn't show significant difference between tillage system at the start of the experiment and in 2015 while in the last year it was significantly higher under CT than NT system. As defined by the USDA, high Bd is an indicator of low soil porosity and soil compaction and it may cause restrictions to root growth, and poor movement of air and water through the soil. According to USDA, the ideal Bd for plant growth for silty soils (as it is the case in this study) is below 1.40 g cm^{-3} and if beyond 1.65 g cm^{-3} it may restrict root growth, and as observed in the tilled plots in 2017 (Table 2.2), the first 30 cm of the soil profile presented values higher than the established ones. Nevertheless, this significant difference between tillage systems may not be plausible, because of the sampling dates. In the previous years, the Bd sampling occurred during the fallow period while in 2017, the samples were taken in the 10th of July under a standing maize crop. Therefore, the high Bd recorded under CT can be caused by the compaction of the soil after ploughing for the seedbed preparation and mechanized traffic while the undisturbed soil has continuous cracks and biopores (old root channels) in the NT system (Martino and Shaykewich, 1994; Huang et al, 2012).

Herein, the soil bulk density is one of the soil parameters that does not follow a defined pattern as some authors reported lower Bd in the soil first 30 cm soil depth under system than minimum tillage (MT) than NT system (Bescansa et al., 2006; Gál et al., 2007). Caparelli Oliveira et al. (2020) also found that Bd was lower in MT than CT and NT while no difference was reported between CT and NT systems in a Typic Hapludult soil in northeast Brazil. Huang et al. (2012) stated that Bd was higher under NT in the first 20 cm soil depth before sowing wheat than under CT however, after harvesting the pattern changed and Bd was higher under CT than under NT and according to these researchers, the rapid increase in Bd through time under the CT system could be caused by the settling of soil after tillage breaking up of the aggregates under the influence of irrigation or rainfall. The results obtained in this study agree with the study of Fuentes et al. (2009) who didn't report significant difference between NT and CT systems at 20 cm soil depth after 14 years in a clay loam soil with maize in Mexico. In another experiment, the soil analysis after 4 years showed no significant differences in Bd between CT and NT systems in a silty clay soil under wheat crop (*Triticum durum* Desf. cv. Aureo) in southern Italy (Ferrara et al, 2017). Ordoñez-Morales et al. (2019) didn't find significant difference between CT and NT systems in a 4-year study in Mexico under a forage oats (*Avena sativa* L.) crop.

Overall, Bd varies according to several factors such as volume and intensity of rainfall, drying and wetting of soil, land position and crop type (Alletto et al., 2009). It depends also on the soil texture and the organic matter particles as well as their packing arrangement.

2.4.2. Soil pH

Based on the results obtained in Table 2.3, the soil pH of this experiment varied from 8.22 to 9.03 and can be classified as moderate to strongly alkaline soil. In 2011, the mean soil pH was significantly higher under NT than CT at 0-30 cm and 0-100 cm soil depth, while in 2015 it was significantly higher under NT system than CT at 0-100 cm soil depth. In 2017, the mean soil pH didn't display significant difference between tillage system at 0-30 cm and was the lowest among all the years. For both NT and CT, soil pH increased with depth (Table 2.3), this may be partially attributed to acidification from N mineralisation in the surface soil layers (Martínez et al., 2004). The results obtained agree with the 6-year study conducted by Lal (1997) who reported a higher soil pH in no till treatments than for plough-based treatments in a continuous maize crop in Nigeria. However, NT system displayed lower pH than CT in an experiment conducted by Limousin and Tessier, (2007) in France on a Luvisol soil and in a maize crop.

The augmentation of the soil pH under NT system in 2015 can be explained by the fact that the soil samples were taken right after harvesting and leaving the crop residues on the surface which can be considered as a new input of OM. According to Pocknee and Sumner (1997), the magnitude of the pH change and the duration of the effect varied with OM type and rate of application, *e.g.*, maize leaves are prone to sum 0.18±0.01 unit to the soil pH immediately after its addition and can reach a maximum pH in 8 days.

2.4.3. Soil organic matter and total nitrogen

In 2017, the SOM was significantly higher under NT system than CT and was also higher than the recorded in 2011 and 2015 at 0-30 cm soil depth (Table 2.4). The non-disturbance of the soil is proven to help the accumulation of organic carbon on the first soil layers as reported by Rasmussen (1999), Lilienfein et al. (2000), During et al. (2002) and Limousin and Tessier (2007). They observed that annual NT and leaving crop residues on the soil surface increase the OM in the topsoil. Lal, 1997 also stated higher OC in non-tilled plots with mulch than in tilled and bare plots.

Considering soil depths, the results obtained in this study showed that most of the SOM content is concentrated in the upper layers of the soil profile especially under NT system (Table 2.4), while there were no significant differences observed all along the soil profile except for 40-60 cm soil depth under CT system. The ploughing of the soil promoted the incorporation of the crop residues and resulted in a homogenous distribution of the SOM all along the soil profile (at least up to 40 cm soil depth) while the absence of the soil tillage in the non-tilled plots helped the accumulation of the SOM in the topsoil (mostly up to 30 cm soil depth). According to the study conducted by Franzluebbers (2002), it is possible to speak about the stratification ratio in this study and which is found to be significantly higher in NT system with 1.17 compared to 0.97 in CT system for 0-10: 20-30 cm soil depth (data not presented). This result is in accordance with the findings of Franzluebbers (2002) who reported a stratification index that varies between 2.0 and 3.4

under NT while it was comprised between 1.1 and 1.9 under CT system. And according to him, a high stratification ratio of C could be good indicator of dynamic soil quality.

The mean equivalent mass of TN didn't show significant differences between tillage systems; however, it was numerically higher under NT system than CT at 0-30 cm and 0-60 cm soil depth (Table 2.5) during all the years of the experiment. In 2017, it can be observed that the mean TN was 0.53 t ha^{-1} higher in the non-tilled plots than in the tilled ones in the first 10 cm of the soil layers, this result agrees with the one reported by Neugschwandtner et al. (2014) as they found that TN was significantly higher under NT system than CT. An 11-year study conducted by Campbell et al. (1996) in Canada confirmed that tillage system had an important influence on the mean TN as it was higher under NT system than CT in 0-7.5 cm soil depth under a durum wheat (*Triticum turgidum* L.). Al-Kaisi et al. (2005) also reported higher TN under NT system than chisel plough treatment in a corn-soybean rotation. Obade et Lal (2014) evaluated the effects of tillage system, in a study conducted in Ohio, on various soil properties under a maize crop and concluded that NT system contributed with more N in the first layer of the soil profile, they also observed that the effects of the tillage system were more significant in the evolution of TN and SOM with them being the main indicators that controlled crop production. Moreover, Xue et al. (2015) found that the reduction of soil disturbance and reduced mineralisation rate of SOM may lead to higher SOC and TN content at 0-5 cm depth under NT.

As shown in Table 2.5, the TN recorded in 2011 was significantly higher than the recorded in 2015 under both tillage systems at 0-60 cm soil depth. This significant difference could be caused by the fact that the vetch crop sowed in 2010 was incorporated in the tilled plots and treated by glyphosate in the non-tilled plots and had a high mineralization rate due to its low C:N ratio which was reflected in the high TN content in 2011 under CT and NT systems.

2.4.4. Nutrient contents: available phosphorus and extractable potassium

In this study, the P content is observed to be higher under CT than NT at 0-30 cm and 0-60 cm soil depth (Table 2.6) for all the years of the study. And as shown, the mean equivalent mass of P is concentrated in the first 10 cm of the soil profile. Neugschwandtner et al. (2014) also found that P accumulation in the reduced tillage (NT

and shallow conservation tillage) occurs in the upper soil layers and depletion in the deepest sampled soil layer over time. Thomas et al. (2007) found that more P was available in the upper (0-10 cm) in NT than in CT in a Luvisol soil under in a semiarid, subtropical environment in Australia. However, López-Fandó and Pardo (2009) reported higher P content in NT than MT and CT soils at 0-20 cm depth, suggesting that the results may have been caused by the accumulation of P in senescent roots. According to Sharpley (1996), topsoil P content is usually greater than that in subsoil due to the sorption of added P and greater biological activity and accumulation of organic material. However, soil P content varies with parent material, extent of pedogenesis, soil texture and managements factors such as soil cultivation and type of applied P.
According to Bradford and Peterson (2000), reduced and NT systems tend to keep soils cooler than clean-tilled systems and since P is a nonmobile nutrient and uptakes depends greatly on root interception, cold soils can create temporary P deficiencies which explains the lower P content under NT compared to CT in this study.

The mean equivalent mass of K didn't show significant difference between tillage systems in 2011 at 0-30 cm and 0-60 cm soil depth and in 2017 at 0-30 cm soil depth, while it was higher under CT than NT in 2015 at 0-60 soil depth. The potassium content presented a stratification index that varied between 1.11 and 1.36 under CT and between 1.34 and 1.65 under NT system for 0-10: 20-30 cm depth (Table 2.7). It can be noticed that the stratification index is higher in the non-tilled plots and this can be explained by the non-disturbance of the soil which promotes an elevated concentration of K in the topsoil while the ploughing of the tilled plots incorporates K and distributes it homogenously all along the soil profile.

Nevertheless, K content tended to be high at deeper soil layers (40-60 cm depth) as it can be observed in Table 2.7 in 2011 and 2015, this can be explained by the fact that K is a mobile ion in soils and consequently significant amounts can be lost by leaching (Alfaro et al., 2004). The soil texture plays an important part in the K leaching, e.g., clayey soils tend to lose 39±0.03 kg ha^{-1} while loamy soils lose about 1- 4 ±0.01 kg ha^{-1} of K (Alfaro et al., 2004); moreover, high amount and increasing depths of irrigation water can influence the leaching of the K applied to the soil and the higher the water depth, the larger the percolated amount of the K^+ ion (Mendes et al., 2016).

2.4.5. Nitrate losses by leaching under CT and NT systems

Nitrate leaching is a prominent process of N loss in agricultural ecosystems and it happens through the nitrification process, the leached nitrate may induce groundwater contamination or surface water eutrophication to threaten human health (Zhang et al.,2015). In this study, the mean NO_3^- concentration varied according to the year and tillage system (Figure 2.4), it was the highest in 2015 under CT and the lowest in 2016 under NT. The variation of this component depended mostly on the amount of water provided during the irrigation months, however, it is necessary to mention that the samples collected for the analysis came from plots that have been fertilized with the highest rate of synthetic fertilizers (FC and FE).

A study conducted by Dowell et al. (1983) found lower levels of NO_3^- in direct drilled soils than in ploughed ones and suggested that nitrification activity was reduced in direct drilled soils relative to ploughed soils which can explain the result obtained in 2015. However, Rice and Smith (1983) found that the rates of nitrification can be higher in NT soils than in CT soils due to more favourable moisture conditions for nitrification in NT soils, which also can be in accordance with the result obtained in 2017.

On one hand and as explained earlier in this study, NT system tend to accumulate more OM in the topsoil which can maintain a constant mineralisation rate all along the crop cycle while ploughing in the tilled soils can stimulate the mineralisation resulting in more N accumulation in the soil (likely to be lost). On the other hand, the N cycle depends on soil temperature, percentage of pores filled with water and oxygen availability which are parameters that can be easily affected by the tillage system.

Quemada et al. (2013) stated that in irrigated agriculture, excessive water application (as it happened in 2015) increase NO_3^- leaching, leading to a vicious circle where low crop N availability is compensated by increasing fertilizer rates and as a consequence, when crops are overwatered, it is common to observe low N efficiency and contamination of the groundwater.

2.5. Conclusion

This study assessed the effect of two tillage systems (CT and NT) on the properties of the soil in a continuous irrigated maize crop to identify practices that can limit the deterioration of the soil quality all over the years. The data presented in our study showed that Bd didn't vary significantly between tillage system in 2011 and 2015 while it was higher under CT than NT in 2017. The changed pattern of Bd in 2017, is mostly caused by the date of sampling and the climatic conditions that differed from the previous years. Mean soil pH was also affected by tillage system at 0-100 cm soil depth in 2011 and 2015 while no difference was noticed in 2017 at 0-30 cm depth, this difference can be mainly explained by the fact that the pH was naturally higher in the NT soils and was accentuated by the addition of crop residues on the soil surface which promoted the OM mineralisation.

After 7 years of non-tilling the soil combined with leaving the crop residues on the soil surface, the SOM presented higher content under NT than CT system especially in the first 30 cm of the soil profile. The mean equivalent mass of the TN did not show significant difference between tillage systems during the three years of the study at 0-30 cm soil depth; however, it was higher in 2011 than in 2015 under CT and NT systems at 0-60 cm soil depth. This could be explained by the fact that incorporating mechanically or chemically the vetch crop in 2010 promoted the fast mineralisation of the SOM and resulted in higher TN in 2011.

Since the start of the experiment, the P content was whether numerically (in 2011 at 0-30 cm and 0-60 cm soil depth) or significantly higher (in 2015 and 2017) under CT than under NT system and was mostly higher in the upper layers of the soil under NT system while it did not vary significantly all along the soil profile under CT system. The content of K did not show significant difference between tillage systems except in 2015, when it was significantly higher under CT system than NT system at 0-60 cm soil depth. It also presented a high stratification index in the first 10 cm layer of the soil under NT system compared to CT system.

An evaluation of the nitrate leaching was also carried out in this study, and the result obtained confirmed that NO_3^- concentration is affected mostly by the amount of water and fertilizer rates provided to the crop while it is likely that the effects of tillage on nitrification depend on specific field conditions, the experiment design and perhaps the synthetic fertilization rates (lack of samples from the different rates of the N fertilization to confirm such statement).

References

Alfaro, M.A., Jarvis, S.C., Gregory, P.J. 2004. Factors affecting potassium leaching in different soils. Soil Use Management, 20: 182-189.

Alletto, L., Coquet, Y. 2009. Temporal and spatial variability of soil bulk density and near-saturated hydraulic conductivity under two contrasted tillage management systems. Geoderm, 152: 85-94.

Al-Kaisi, M.M., Yin, X., Licht, M.A. 2005. Soil carbon and nitrogen changes as affected by tillage system and crop biomass in a corn-soybean rotation. Applied Soil Ecology, 30: 174-191.

Allison, L.E., Moodie, C.D. 1965. Carbonate. In Black, C.A. (eds.) Methods of soil analysis, part 2. Agronomy, 9:1379-1396.

Baxter, N., Wiliamson, J. 2001. Know your soils. Part 1 Introduction to Soils. Agriculture Victoria. Centre for Land Protection Research, pp28.

Bescana, P., Imaz, M, J., Virto, I., Enrique, A., Hoogmoed, W.B. 2006. Soil water retention as affected by tillage and residue management in semiarid Spain. Soil & Tillage Research, 87: 19-27.

Blake, G.R. 1965. Bulk density. In: Black, C.A. (Ed.), Methods of Soil Analysis. Part I. Physical and Mineralogical Properties. SSSA Inc., Madison, WI, USA, pp. 374–390.

Bradford, J. M., Peterson, G. A. 2000. Conservation tillage. In Handbook of soil science, ed. M. E. Sumner, G247-G269. Boca Raton, FL, USA: CRC Pres

Bronick, C.J., Lal, R. 2005. Soil structure and management: a review. Geoderma, 124: 3-22.

Bundy, L.G., Bremner, J.M. 1972. Simple titrimetric method for determination of inorganic carbon in soils. Soil Science Society of America Journal, 36(2): 273-275.

Cambardella CA and Elliot ET. 1992. Particulate soil organic matter changes across a grassland cultivation sequence. Soil Science Society Am J 56:777-83.

Campbell, C.A., McConkey, B.G., Zentner, R.P., Selles, F., Curtin, D. 1996. Tillage and crop rotation effects on soil organic C and N in a coarse-textured Typic Haploboroll in southwestern Saskatchewan. Soil & Tillage Research, 37: 3-14.

Caparelli Oliveira, F.C., Dias Ferreira, G.W., Santos Souza, J.L., Oliveira Vieira M.E., Pedrotti, A. 2020. Soil physical properties and soil organic carbon content in northeast Brazil: long-term tillage systems effects. Scientia Agricola 77(4): 1-6.

Dam, R.F., Mehdi, B.B., Burgess, M.S.E., Madramootoo, C.A., Mehuysa, G.R., Callum, I.R. 2005. Soil bulk density and crop yield under eleven consecutive years of corn with different tillage and residue practices in a sandy loam soil in central Canada. Soil & Tillage Research, 84: 41-53.

Doran, J.W., Parkin, T.B. 1994. Defining and assessing soil quality. Defining soil quality for a Sustainable Environment, 35: 3-21.

Doran, J.W., Parkin, T.B. 1996. Quantitative indicators of soil quality: a minimum data set. In J.W. Doran and A.J. Jones, eds. Methods for Assessing Soil Quality. SSSA, Inc., Madison, Wisconsin, USA.

Dowell, R.J., Crees, R., Cannell, R.Q. 1983. Afield study of contrasting methods of cultivation on soil nitrate content during autumn and spring. Journal of Soil Science, 34:367-379.

During, R.A., Thorsten, H., Stefan, G. 2002. Depth distribution and bioavailability of pollutants in long-term differently tilled soils. Soil & Tillage Research, 66: 183-195.

Ellert, B. H., Bettany, J. 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Can. J. Soil. Sci. 75: 529-538.

Ferraraa, R.M., Mazza, G., Muschitiello, C., Castellini, M., Stellacci, A.M., Navarro, A., Lagomarsino, A., Vitti, C., Rossi, R., Rana, G. 2017. Short-term effects of conversion to no-tillage on respiration and chemical - physical properties of the soil: a case study in a wheat cropping system in semi-dry environment. Italian Journal of Agrometeorology, 22(1):47- 58.

Franzluebbers, A.J. 2002. Soil organic matter stratification ratio as an indicator of soil quality. Soil & Tillage Research, 66: 96-106.

Fuentes, M., Govaerts, B., De Leon, F., Hidalgo, C., Dendooven, L., Sayre, K.D., Etcheveres, J. 2009. Fourteen years of applying zero tillage and conventional tillage, crop rotation and residue management systems and its effect on physical and chemical soil quality. European Journal of Agronomy, 30: 228-237.

Gál, A., Vyn, T.J., Michéli, E., Kladivko, E.J., McFee, W.W. 2007. Soil carbon and nitrogen accumulation with long-term no-till versus mouldboard plowing overestimated with tille-zone sampling depths. Soil & Tillage Research, 96; 42-51.

Gozubuyuk, Z., Sahin, U., Ozturk, I., Celik, A., Adiguzel, M.C. 2014. Tillage effects on certain physical and hydraulic properties of a loamy soil under a crop rotation in a semi-arid region with a cool climate. Catena, 118: 195-205.

Griffis, T.J., Lee, X., Baker, J. M., Russelle, M. P., Zhang, X., Venterea, R., Millet D. B. 2013. Reconciling the differences between top-down and bottom-up estimates of nitrous oxide emissions for the U.S. Corn Belt. Global Biogeochemical Cycles, 27: 746-754.

Huang, G.B., Chai, Q., Feng, F.X., Yu, A.Z. 2012. Effects of Different Tillage Systems on Soil Properties, Root Growth, Grain Yield, and Water Use Efficiency of Winter Wheat (*Triticum aestivum* L.) in Arid Northwest China. Journal of Integrative Agriculture, 11(8): 1286-1296.

Huang, M., Liang, T., Wang, L., Zhou, C. 2015. Effects of no-tillage systems on soil physical properties and carbon sequestration under long-term wheat–maize double cropping system. Catena, 128: 195-202.

Jarecki, M.K., Lal, R. 2003. Crop management for soil carbon sequestration. Critical Reviews in Plant Science, 22 (6):471-502.

Kaiser, J. 2004. Wounding Earth’s Fragile Skin. Science 11(304): 1616-1618.

Kalev, S.D., Toor, G.S. 2018. The composition of Soils and Sediments. Green Chemistry. An Inclusive Approach: 339-357.

Karlen, D.L., Mausbach, M.J., Doran, J.W., Cline, R.G., Harris, R.F., and Schuman, G.E. 1997. Soil quality: A concept, definition, and framework for evaluation. Soil Science Society of American Journal, 61:4-10.

Lal, R. 1997. Long-term tillage and maize monoculture effects on a tropical Alfisol in western Nigeria. II. Soil chemical properties. Soil & Tillage Research, 42: 161-174.

Lal, R. 2009. Soil quality impacts of residue removal for bioethanol production. Soil & Tillage Research, 102: 233-241.

Lilienfein, J., Wilcke, W., Vilela, L., Do Carmo Lima, S., Thomas, R., Zech, W. 2000. Effect of no-tillage and conventional tillage systems on the chemical composition of soil solid phase and soil solution of Brazilian savanna oxisols. J. Plant Nutr. Soil Sci. 163, 411–419.

Limousin, G., Tessier, D. 2007. Effects of no-tillage on chemical gradients and topsoil acidification. Soil & Tillage Research, 92: 167-174.

López-Fandó, C., Pardo, M.T. 2009. Changes in soil chemical characteristics with different tillage practices in a semi-arid environment. Soil & Tillage Research, 104: 278-284.

Martino, D. L., Shaykewich, C. F. 1994. Root penetration profiles of wheat and barley as affected by soil penetration resistance in field conditions. Canadian Journal of Soil Science, 74: 193-200.

Martinez, M.J., Farsaoui, K., de Prado, R. 2004. Weed control and persistence of two oxyfluorfen formulations in olive groves under non tillage conditions. Communications in Agricultural and applied Biological Sciences, 69(3):77-82.

Mendes, W.C., Júnior, J.A., da Cunha, P.C.R., da Silva, A.R., Evangelista, A.W.P., Casaroli, D. 2016. Potassium leaching in different soils as function of irrigation depths. Revista Brasileira de Engenharia Agrícola e Ambiental, 20(11): 972-977.

Mrabet, R., Saber, N., El-Brahli, A., Lahlou, S, Bessam, F. 2001. Total, particulate organic matter and structural stability of a Calcixeroll soil under different wheat rotations and tillage systems in a semiarid area of Morocco. Soil & Tillage Research, 54(4): 225-235.

Nelson, L.M. 1983. Hydrogen recycling by Rhizobium leguminosarum isolates and growth and nitrogen contents of pea-plants *(Pisum-sativum* L). Applied and Environment Microbiology, 45(3): 856-861.

Neugshwandtner, R.W., Liebhard, P., Kaul, H.P., Wagentristl, H. 2014. Soil chemical properties as affected by tillage and crop rotation in a long-term field experiment. Plant Soil Environment, 60(2) : 57-62.

NRCS. 2006. Natural Resources Conservation Service, Soils- Fundamental Concepts. National Soil Survey Centre, Lincoln, NE.

Obade, V. d. P., Lal, R. 2014. Soil quality evaluation under different land management prctices. Environmental Earth Sciences, 72(11):

Olsen, S. R. and Sommers, L. E. 1982. Phosphorus. pp. 403-430. *In* : A. L. Page, et al. (eds.) Methods of soil analysis: Part 2. Chemical and microbiological properties. Agron. Mongr. 9. 2nd ed. ASA and SSSA, Madison, WI.

Ordoñez-Morales, K.D., Cadena-Zapata, M., Zermeño-González, A., Campos-Magaña, S. 2019. Effect of Tillage Systems on Physical Properties of a Clay Loam Soil under Oats. Agriculture, 9(3): 62.

Pimentel, D., Burgess, M. 2013. Soil Erosion Threatens Food Production. Agriculture, 3: 443-463.

Pocknee, S., Sumner, M.E. 1997. Cation and Nitrogen Contents of Organic Matter Determine Its Soil Liming Potential. Soil Science and Society of American Journal, 61:89- 92.

Quemada, M., Baranski, M., Nobel-de Lange, M.N.J., Vallejo, A., Cooper, J.M. 2013. Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. Agriculture, Ecosystems and Environment, 174: 1-10.

Rasmussen, K.J. 1999. Impact of ploughless soil tillage on yield and soil quality: A Scandinavian review. Soil & Tillage Research, 53: 3-14.

Reynolds, M., Tuberosa, R. 2008. Translational research impacting on crop productivity in drought prone environments. Current Opinion in Plant Biology, 11:171-179.

Rice, C.W., Smith, M.S. 1983. Nitrification of Fertilizer and Mineralized Ammonium in No-Till and Plowed Soils. Soil Science and Society of America Journal, 47:1125-1129.

Salem, H.M., Valero, C., Muñoz, M.A., Gil Rodríguez, M., Silva, L.L. 2015. Short-term effects of four tillage practices on soil physical properties, soil water potential, and maize yield. Geoderma, 237-238: 60-70.

Sharpley, A. 1996. Availability of of residual phosphorus in manured soils. Soil Science Society of America Journal, 60: 1459-1466.

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

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S. 2015. Planetary boundaries: Guiding human development on a changing planet. Science, 347: 6223.

Tolimir, M., Kresović, B., Životić, L., Dragović, S., Dragović, R., Sredojević, Z., Gajić, B. 2020. The conversion of forestland into agricultural land without appropriate measures to conserve SOM leads to the degradation of physical and rheological soil properties. Science Reports 10, 13668.

Thomas, G.A., Dalal, R.C., Standley, J. 2007. No-till effects on organic matter, pH, cation exchange capacity and nutrient distribution in a Luvisol in the semi-arid subtropics. Soil & Tillage Research, 94: 295-304.

Verhulst, N., Govaerts, B., Verachtert, E., Castellanos-Navarrete, A., Mezzalama, M., Wall, P., Deckers, J., Sayre, K.D., 2010. Conservation Agriculture, Improving Soil

Quality for Sustainable Production Systems? In: Lal, R., Stewart, B.A. (Eds.), Advances in Soil Science: Food Security and Soil Quality. CRC Press, Boca Raton, FL, USA, pp. 137-208.

United Nations. 2021. https://www.un.org/en/sections/issues-depth/population/index.html.

Xue, J.F., Pu, C., Liu, S.L., Chen, Z.D., Chen, F., Xiao, X.P. 2015. Effects of tillage systems on soil organic carbon and total nitrogen in a double paddy cropping system in Southern China. Soil & Tillage Research, 153: 161-168.

Walkley, A.J., Black, I.A. 1934. Estimation of soil organic carbon by the chromic acid titration method. Soil Science, 37: 29-38.

Zhang, X., Davidson, E.A., Mauzerall, D.L., Searchinger, T.D., Dumas, P., Shen, Y. 2015. Managing nitrogen for sustainable development. Nature, 528: 51.-59.

CHAPTER 3. INFLUENCE OF TWO TILLAGE SYSTEMS ON WATER DYNAMICS AND GRAIN YIELD OF AN IRRIGATED MAIZE IN A SEMIARID AREA OF CASTILE AND LEON, SPAIN

3.1. Introduction

It is predicted that by 2050, the global population will reach more than 9 billion people and will lead to an increase of the demand for high-quality food and the competition for increasingly scarce land, water and energy resources is intensifying with particular pressure on agriculture (FAO, 2017). This constant increase of the population and the high climate extremes are challenging agriculture to maintain a sufficient level of crop production while reducing, especially, the amount of water used, therefore increasing its use efficiency (Kijne et al., 2003) mainly for major crops like maize (*Zea mays* L.). In fact, maize is one of the most important crops worldwide and hold the third place behind rice and wheat (WMO, 2012). Its production plays a major role in the economic and socio-economic development of many countries (Greaves and Wang, 2017b), such as Spain, and where the community of Castile and Leon is the second important producer of maize grain with a production of 940.561 tons in 2017 (MAPAMA, 2017).

The climatic conditions in Castile and Leon (north-western Spain) are characterized by long cold periods that extend almost continuously for much of the year and high temperatures during summer. Moreover, due to the mountainous barriers that surround this region, precipitations are very scarce and are unequally distributed. These conditions are challenging for maize, because of its high sensitivity to drought, which can cause severe yield reduction. Under such climatic conditions, maize productivity depends to a high extent on irrigation supplies. Stone et al., (1996) confirmed that optimum grain yield have high irrigation demands. The lack of water and/or its irregularity is particularly damaging to grain yield if it occurs in the growing season, at flowering, and during mid to late grain filling (Heisey and Edmeades, 1999).

The water applied to the crop infiltrates the soil, some is absorbed by plants (and later lost through transpiration), some percolates more deeply, recharging groundwater (FAO, 2004), and the excess water drains through the vadose zone into the groundwater, which contributes to aquifer recharges. In this context, the soil plays the role of a temporal water reservoir that transforms an irregular precipitation (irrigation and/or rainfall) into a continuous known source, hence supplying moisture to the root zone (Fares and Alva, 2000). This implies that soil physical properties exert a dominant effect on soil-water balance, crop production and water use efficiency especially in regions prone to drought stress (Lal, 1991). Therefore, continuous monitoring of soil water content (SWC) within and below the rooting zone can facilitate optimal irrigation scheduling aimed at minimizing both the effects of water stress on plants and the leaching of water below the root zone (Fares and Alva, 2000).

Besides the climatic conditions and the availability of accessible water, the SWC depends on other factors such as tillage systems. Result of various investigations from almost all world climatic zones suggest that ploughing causes common soil-related problems of compaction, erosion, reduced water percolation and thus increased runoff and high energy and time requirements (Titi, 2003). While conservation tillage, and other system with at least 30 percent residue cover remaining after planting (Derpsch, 2001), is generally designed to reduce soil erosion (Reinbott et al. 2004), decrease runoff, increase infiltration rate and lessen the evaporation of the soils' water (Arshad et al., 1999; Rasmussen, 1999). According to Lampurlanés et al. (2001), conservation tillage increases stored soil water by increasing infiltration and reducing evaporation, in semi-arid conditions of the Ebro Valley (north-eastern Spain), but depending on the soil type and climatic conditions, this leads to higher, equal or even lower yields than conventional tillage (CT) systems. Some authors reported that grain yields were always equal or higher in no-tillage (NT) system than on mouldboard ploughed plots (Lal et al., 1978; McMaster et al., 2002; De Vita et al., 2007; Copec et al., 2015). While others found that NT grain yields were lower than those of CT practice (Chopart and Kone, 1985; Wilhelm et al., 1987; Guzha, 2004). Therefore, it is important to study the effect of soil tillage or its absence on the SWC and grain yield in the community of Castile and Leon where only few studies were conducted. The fact that maize roots tend to explore the lower soil profile to a greater extent under water deficit and with the conservation of soil water content under NT system may allow an increase of water productivity (Lamm et al., 2009). Moreover, water use efficiency (WUE) and irrigation water use efficiency (IWUE) are important indicators for evaluating the water-saving efficiency of irrigated field crops (Greaves and Wang, 2017a; Kang et al., 2017). Stone et al. (2001) reported a water use (WU) of 311 mm and 98 mm for fully irrigated and drought treatments, respectively and found that early droughts increased the WUE when compared with late drought treatments. In contrast, Payero et al. (2008) recorded a maize grain yield of 844 and 1040 g m^{-2} when WUE was equal to 1.5 and 1.6 kg m^{-3}, for deficit and fully irrigated maize, respectively. Yazar et al. (1999) also found the highest yield, WUE and dry matter under fully irrigated treatment and under 80 % of the estimated irrigation requirement, when they studied the effect of six different irrigation levels. In a study by Liu et al. (2011), WUE was 1.6 kg m^{-3} when maize grain yield reached 9.5 t ha^{-1}. In 2008, Kresović et al. (2016) reported maize yield of 8.7 and 14.3 t ha^{-1} corresponding to WUE of 2.9 and 2.8 kg m^{-3}, respectively. In this context, farmers need to manage the water resources and adopt appropriate tillage practices without removing residues in order to effectively store and use the limited amount of precipitation and water from irrigation for crop production and to control soil erosion. In addition, a good understanding of WUE and IWUE is important to assess the financial benefits of irrigation strategies and minimize the economic impact.

Furthermore, the assessment of water productivity according to tillage systems was scarcely established in the region of Castile and Leon. Therefore, it was interesting to study the influence of CT and NT managements on the SWC and water balance at different soil depths (50, 100 and 150 cm), grain yield and water productivity from 2015 to 2017 in irrigated maize crop in the semi-arid condition of Castile and Leon, Spain.

3.2. Material and methods

3.2.1. Experimental design and crop management

To avoid repetition, the experimental design and crop management methodology are detailed in ***Chapter 3***.

To determine the grain yield and its components, plant samples were picked in one-meter area from four rows from each tillage system. Afterwards, ear numbers per sample, rows per ear and grain numbers per row were counted and the grain yield was estimated.

3.2.2. Determination of the soil water content

The soil in the experimental site was characterized by a silty loam texture and the main properties determined in this study are shown in Table 3.1. The field capacity (FC) and the permanent wilting point (PWP) were determined for every 10 cm throughout 150 cm soil depth based on the water retention curve (pF curve) using the pressure chamber developed by Richards (1941). The apparatus used was formed by a hermetic pressure chamber of steel, where the soil samples were placed in cylinders of a known volume. At its base, a semipermeable ceramic plate was deposited according to the potential to be determined and which allowed the extracted water from the samples to drain through a collector. After that, the soil samples were placed to dry at 105ºC then weighed to determine the hydric content.

Table 3.1: Soil physical properties at the experimental site in 50, 100 and 150 cm soil depth under conventional tillage (CT) and no-tillage (NT) system.

Soil depth (cm)	50		100		150	
Tillage system	CT	NT	CT	NT	CT	NT
Bulk density (g cm^{-3})	1.50	1.50	2.0	1.7	1.6	1.5
Texture (%)						
Sand	34.0	31.0	31.0	27.0	29.0	33.5
Silt	46.0	49.0	45.0	50.0	47.0	45.5
Clay	19.0	19.0	23.0	23.0	25.0	21.0
pH	8.6	8.7	8.8	8.9	9.0	9.1
FC (%)	28.9		28.8		27.8	
PWP (%)	12.3		10.8		14.4	
Available water (%)	16.6		18.0		13.3	

A capacitance probe model Diviner 2000 (Sentek Pty LTd, Adelaide, SA.) was used in this experiment. The sensor is a hand-held, portable soil moisture monitoring device consisting of a display/ logger unit, connected by a cable to an automatic depth-sensing probe that is moved up and down in an access tube. During calibration, the manufacturer recommends normalizing the probe using the scaled frequency (SF) values for air and water (≈ 25ºC).

$$SF = \frac{Fa - Fs}{Fa - Fw}$$

Where: F_a is the SF in the air; F_s is the SF in soil; F_w is the SF in water.

The normalization is necessary to obtain meaningful data continuously. The values for F_a and F_w were fixed at 176900 and 127591, respectively.

During all the crop seasons and after the maize sowing, six PVC plastic access tubes were installed in both CT and NT treatments, following the procedures suggested by the manufacturer (Sentek, 2000) to ensure good contact between the soil and the access tube wall. The tubes were 1.5 m long with a diameter of 50.8 mm. The access tubes were driven into soil using a sledgehammer, extracting the soil inside the tube with a 47 mm soil auger, and being careful not to empty the augured soil onto the surrounding site to avoid any change of the infiltration rate of rainfall and irrigation. A double-ring rubber plug was installed at the bottom of each tube to avoid water and/ or vapour entering into the access tube. After the installation, a small part of the access tubes was left above the soil surface to prevent water entrance. A plastic top cap was firmly fitted to the upper end of each access tube. Throughout the crop season, the measurements were accomplished by inserting the probe in the PVC tube. The probe takes two measurements of SWC per 10 cm, one during descending and one during ascending. The average of these measurements is stored in the probe data logger (Paraskevas et al., 2012). To generate absolute data, the manufacturer (Sentek, 2000), delivers a default equation already calibrated, but not suitable for all soil types. Therefore, it is recommended that soil-specific calibrations should be conducted. For this purpose, soil samples were taken to determine the water content using the gravimetric method and bulk density using the cylinder method. Then, the volumetric water content was estimated using the equation suggested by Haberland et al. (2015):

$\theta_w = WBd$

Where θ_w *is* the volumetric water content; W is the gravimetric water content and Bd is the bulk density.

A regression analysis was conducted on the values of volumetric water content and the normalised frequencies provided by the capacitance probe, in order to obtain calibration equations. The results obtained during the 3-year study did not show significant correlation between the volumetric content and the measurements given by the probe, therefore, the equation defined by Groves and Rose (2004) for silty clay loam soil was used to estimate the SWC:

$$SF = 0.3531\, \theta_w^{0.2621}$$

where SF is the scaled frequency (dimensionless) and θ_w is the volumetric water content (%).

3.2.3. Soil water balance

The crop evapotranspiration differs distinctly from the reference evapotranspiration (ET_0) as the ground cover, canopy properties, and aerodynamics resistance of the crop are different from grass. The effects of characteristics that distinguish field crops from grass are integrated into the crop coefficient (K_c). The crop evapotranspiration is calculated using the following equation:

$ETc = K_c\, ET_0$

Where ETc is the crop evapotranspiration (mm d^{-1}); K_c is the crop coefficient; ET_0 is the reference crop evapotranspiration (mm d^{-1}).

Most of the effects of the various weather conditions are incorporated into the ET_0 estimate. Therefore, as ET_0 represents an index of climatic demand, Kc varies predominately with the specific crop characteristics and only to a limited extent with climate. The reference ET_0 is defined and calculated using the FAO Penman-Monteith equation (Allen et al., 1998). To define the K_c maize coefficient in this study; K_c values estimated by the meteorological station and those obtained by the FAO were taken into account; both were expressed in function of the maize development stages in the field. The variation of the crop coefficient K_c is due to the changes in vegetation and ground cover and to differences in evapotranspiration during the growing period. This growing season can be divided into distinct growth stages: initial, crop development, mid-season and late season.

The water balance using the actual daily crop evapotranspiration (ET_c) was calculated by the model given in the following equation:

$$ETc = P + I - D - R \pm \Delta W$$ (Kuscu et al.,2013)

where P is the rainfall; I the irrigation; D the drainage; R the run-off and ΔW is the change in soil water storage. All terms are expressed in mm of water. D was estimated using the previous equation. R was assumed zero because the irrigation applied did not cause runoff. ΔW was estimated from measured soil moisture content obtained by the capacitance probe (Diviner 2000).

3.2.4. Assessment of water productivity

The water productivity concepts used are those defined by Garcia y Garcia et al. (2009) and Greaves and Wang (2017a). The total water productivity, also known as the water use efficiency (WUE, kg m^{-3}) and the irrigation water productivity (IWUE, kg m^{-3}) was estimated as:

$$\mathrm{WUE} = \frac{Y_a}{\mathrm{WU}}; \quad \mathrm{IWUE} = \frac{Y_a}{I}$$

where Y_a is the actual grain yield (t ha^{-1}) achieved for both tillage systems; WU is the sum of ETc from planting to maturity (mm) and I is the irrigation (mm).

3.2.5. Statistical analysis

Comparison of all the data collected from the SWC measurements, water balance and grain yields of both CT and NT systems during the 3-year study was performed by analysis of variance (ANOVA) in Infostat, statistical software. Treatment means were separated using the Tukey test at the 5% significant level ($P \leq 0.05$).

3.3. Results

3.3.1. Weather conditions

Table 3.2: Monthly rainfall and mean temperature during the 3-year study and 33 years mean (1981-2014) at Zamadueñas experimental field, Spain.

Month	2015			2016			2017			1981-2014	
	T_{mean} (°C)	ETc (mm)	Rainfall (mm)	T_{mean} (°C)	ETc (mm)	Rainfall (mm)	T_{mean} (°C)	ETc (mm)	Rainfall (mm)	T_{mean} (°C)	Rainfall (mm)
April	12.1	-	-	9.6	5.7	2.0	12.5	-	-	10.9	-
May	15.8	60.4	19.8	13.3	51.7	47.4	16.8	59.6	42.0	14.5	37.5
June	19.9	101.2	76.2	19.0	106.5	1.9	22.5	114.2	5.4	18.6	23.8
July	23.7	231.7	4.2	23.2	224.7	5.4	22.2	225.7	33.2	21.5	20.7
August	21.3	154.0	5.2	22.5	155.1	0.2	22.0	163.7	13.6	21.6	16.2
September	16.8	46.9	23.6	19.0	49.1	13.0	17.7	53.9	0.2	18.5	25.9
October	13.3	25.8	54.2	14.5	25.7	46.2	15.6	14.1	-	13.6	46.3
November	9.1	10.0	46.8	7.3	4.1	25.0	7.0	-	-	7.9	50.9
Total/ Mean	16.5	629.9	230.0	16.0	622.6	141.1	17.0	631.1	94.4	15.9	221.2

Table 3.2 shows the weather conditions recorded at the local meteorological station during the studied period. As is can be noticed, the experiment site is characterized by low precipitation and high temperature during the maize growing season. The first year of the study (2015) displayed higher precipitation and mean temperature than the ones recorded in the 33-years mean. However, these precipitations witnessed high irregularity throughout the maize growing season with the lowest precipitation in July and the highest in June, it is also necessary to mention that during this month there were days when high and strong rainfall occurred in short periods. The second year of the study (2016) presented lower precipitation than the 33 years mean and showed high precipitation irregularity as well throughout the period of the growing season with the highest amount in May and the lowest in August. The last year (2017) was characterized by long drought

period with the lowest precipitation and the highest temperature recorded during the study and in comparison, with the 33 years mean. In addition to the low precipitations that characterize the experimental site, it can be observed in Table 3.2 that ETc was generally high during the growing season of maize. The highest ETc corresponded to the month of July followed by August during the three years which actually coincides when the tasseling stage and the start of the grain filling happen and made it very indispensable to realize irrigation treatments to meet the hydric needs of the crop in order to assure a decent grain yield production.

3.3.2. Soil water dynamics

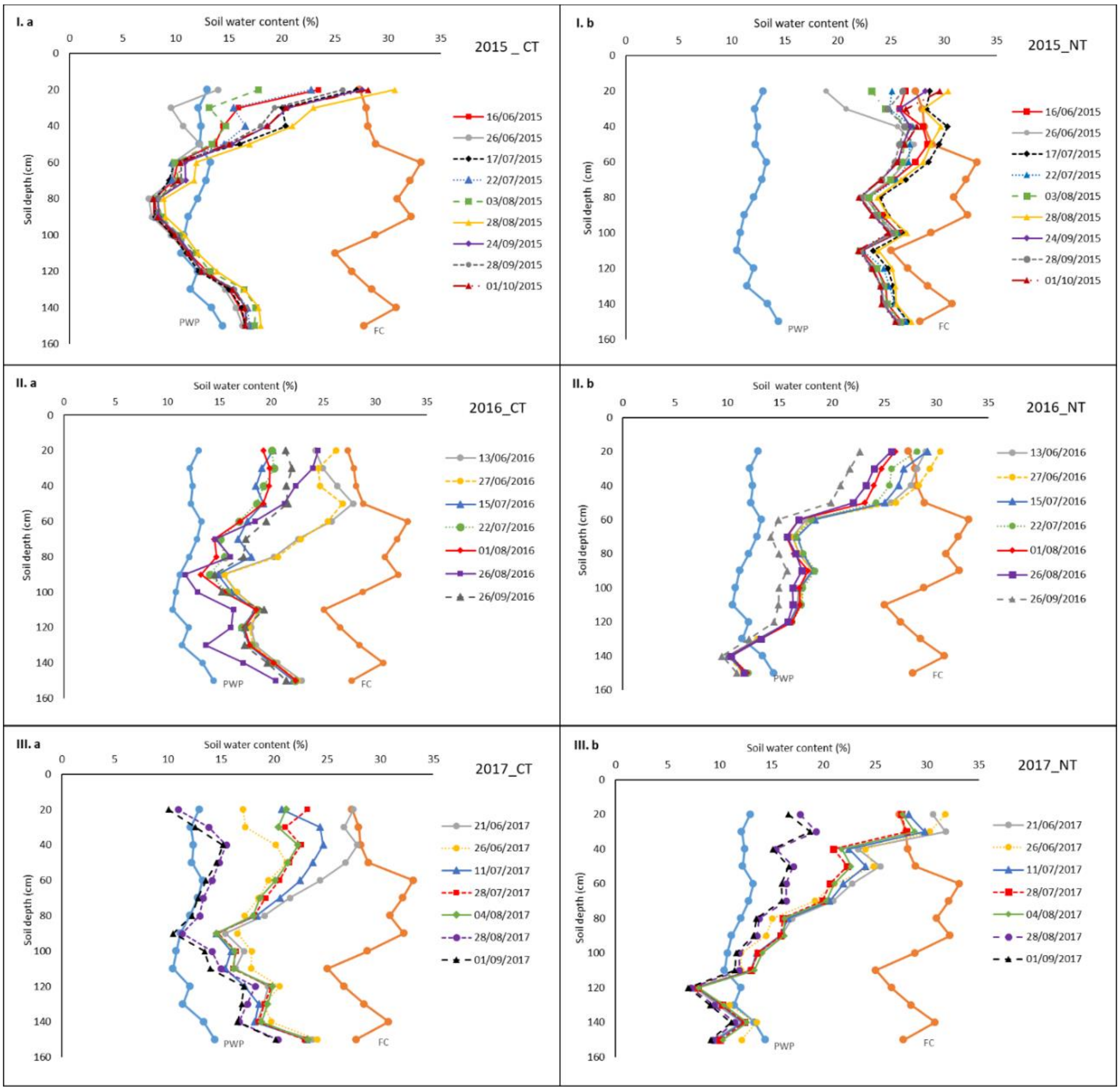

Figure 3.1. Soil water dynamic through the soil profile at different dates of the maize growing seasons under conventional (CT) and no tillage (NT) systems in 2015, 2016 and 2017. PWP, permanent wilting point; FC, field capacity.

The PWP recorded at the start of the experiment varied from 12.3 % at 10 cm to 14.4 % at 150 cm while the FC reached 28.9 and 27.8 % at the same depths respectively (Table 3.1). Both of these factors were used to locate the seasonal variation of the SWC of the maize crop during the growing season (Figure 3.1). As it can be observed in Figure 3.1, generally the SWC measurements recorded were comprised between the PWP and FC during the 3 years of the study and under both tillage systems except in 2015 when the SWC displayed lower percentages than the PWP at 60-100 cm soil depth under CT management. Moreover, the SWC recorded in 2016 and 2017 under NT system at 130-150 cm and at 110-150 cm soil depth respectively was lower than PWP. In 2015, the measurements recorded at the first 20 cm under CT system (Figure 3.1.I. a) were higher than PWP (12.9%) and closer to FC (27.4%) at the same depths and ranged from 13.9 to 30.7 %. From 30 cm depth, these values started to decrease and the SWC reported from 60 cm to 100 cm soil depth presented lower percentages than PWP recorded at the same depths and came back to higher values at 120 cm. While under NT practice (Figure 3.1.I. b), the SWC values were closer to FC and did not present high variation all along the soil profile. It is also necessary to mention that the measurements taken on the 26th of June 2015 displayed lower values than PWP at 30-50 cm soil depth under CT system and were the lowest at 20-30 cm soil depth under NT practice (Figure 3.1. I. a, b). In 2016, the SWC displayed percentages that were comprised between PWP and FC all along the soil profile and varied according to the measurement dates under CT system (Figure 3.1. II. a). However, in NT plots, the SWC was closer to the FC at 20-40 cm depth with percentages that ranged from 20.8 to 30.4% but started decreasing at 50 cm to get closer to the PWP until 130 cm depth where the SWC recorded lower percentages than the PWP at the same depths (Figure 3.1. II. b). In 2017, the SWC reported was generally comprised between the PWP and FC, except for the last measurements on August, 28 and September 1st when the SWC displayed lower percentages than PWP at 20 and 90 cm depth under CT system. It is also observed that the SWC varied among the dates of the measurements in the first 80 cm depth (Figure 3.1. III. a). While, NT practice recorded higher SWC than the FC at 20-30 cm depth and ranged from 27.5 to 31.8 % except for the last two measurements that displayed percentages comprised between both PWP and FC, however a gradual SWC decrease was observed starting at 30 cm and reached lower values than the PWP at 110 cm depth (Figure 3.1. III. b) for all the measurements dates.

Table 3.3. The cumulative soil water content (mm) during the maize reproductive growth stages in the 3-year study under conventional tillage (CT) and no-tillage (NT).

Time of season	Soil depth (cm)	**2015**		**2016**		**2017**	
		CT	NT	CT	NT	CT	NT
Tasseling in July	50	83.5	117.0	77.0	107.5	93.4	104.6
	100	128.7	247.1	160.6	195.1	185.4	193.2
	150	199.7	372.4	256.2	262.8	277.9	246.3
Silking in July	50	69.3	103.9	78.2	103.7	90.2	95.8
	100	114.8	228.8	155.5	191.2	180.1	180.7
	150	187.8	352.2	251.8	260.2	275.0	233.2
Blister in early August	50	59.1	100.6	78.1	98.1	88.2	98.9
	100	106.5	223.7	153.1	181.6	177.2	185.2
	150	183.2	344.3	249.6	250.2	273.6	239.0
Milk in August	50	70.3	103.0	79.4	96.2	85.1	101.0
	100	117.6	226.2	158.9	178.9	172.9	188.9
	150	193.8	348.5	255.2	247.6	270.4	243.8
Dough in mid-August	50	87.3	116.8	73.0	92.7	77.2	92.9
	100	135.0	246.2	150.3	172.6	157.2	175.7
	150	210.3	373.3	247.4	239.6	249.1	227.6
Dent in late-August	50	91.6	117.0	92.1	95.1	55.1	69.7
	100	144.1	246.0	165.8	177.7	121.1	141.9
	150	222.5	372.9	249.7	245.0	208.9	191.4
Physiological Maturity in September	50	83.3	108.9	86.4	85.2	52.4	67.2
	100	131.1	231.3	170.6	159.9	115.1	137.7
	150	204.7	352.8	265.8	221.8	200.0	185.7
Mean	50	77.8 b	109.6 a	80.6 b	96.9 a	77.4 b	90.0 a
	100	125.4 b	235.6 a	159.3 a	179.6 a	158.4 a	171.9 a
	150	200.3 b	359.5 a	253.7 a	246.8 a	250.7 a	223.8 b

Tillage system		**CT**	**NT**
Mean	50	78.6 b	98.8 a
	100	147.7 b	195.7 a
	150	234.9 b	276.7 a

Mean values followed by different letters are significantly different between tillage system ($P \leq 0.05$).

In 2015 (Table 3.3), the cumulative SWC (CSWC) varied during the maize reproductive growth stages under both tillage systems and at all depths. The SWC did not show significant differences among the different stages however it numerically displayed the highest values during the dent stage under both tillage systems, followed by the dough, the tasselling and the physiological maturity stages, while the lowest water content was attributed to the blister and the silking stages. In addition, NT system presented significantly higher SWC that was 40, 88 and 80 % than CT management at 50, 100 and 150 cm soil depth respectively.

In 2016 (Table 3.3), the CSWC also varied during the reproductive growth stages of maize without displaying significant differences, however it presented the highest values during the month of July which corresponded to both tasselling and silking stages under NT system while under CT management it reached its peak in late August (dent) and September (physiological maturity). It is also observed that NT system displayed a CSWC 20% significantly higher than CT in the first 50 cm while no significant differences were reported for both 100 and 150 cm under both tillage managements although, NT displayed higher CSWC values during the two months of July and August until the 23rd of September where CT recorded higher water content.

The CSWC recorded during the last year of the trial (Table 3.3) did not display significant differences among the growth stages of the crop, but it presented the highest value during the tasselling in July under both tillage systems, then it can be observed that it decreased gradually during the following stages. It is also necessary to mention that during this year (2017) the mean cumulative SWC recorded at 50 cm soil depth was 16% significantly higher under NT system than CT while at 150 cm this parameter was 12% higher under CT than NT management.

According to Table 3.3, the mean cumulative SWC was significantly higher under NT management than CT treatment. In fact, in a long-term period NT system presented 11, 14 and 8% more cumulative SWC than CT system at 50, 100 and 150 cm depths, respectively.

Table 3.4. Water balance according to conventional tillage (CT) and no-tillage (NT) system during the 3- years study at 50, 100 and 150 cm depth.

Year	Month	Etc (mm)	Rainfall (mm)	Irrigation (mm)	Tillage system	ΔW50	D50	ΔW100	D100	ΔW150	D150
						(mm)					
2015	June	69.5	7.6	36	CT	24.9	-50.8	-24.1	-50.0	-24.9	-50.8
					NT	-26.4	-52.4	-27.8	-53.7	-28.1	-54.0
	July	231.7	4.2	288	CT	31.5	92.0	37.5	98.0	42.0	102.5
					NT	25.8	86.3	29.3	89.9	31.8	92.3
	August	154.0	5.2	223	CT	20.8	94.7	22.4	96.4	23.3	97.2
					NT	6.2	80.1	8.5	82.5	11.4	85.4
	September	46.9	23.6	117	CT	-8.7	85.0	-12.4	81.3	-16.2	77.5
					NT	-15.2	78.5	-21.9	71.9	-28.4	65.3
2016	June	77.9	0.6	81	CT	-1.5	2.2	0.8	2.9	-1.9	1.8
					NT	4.7	8.4	5.6	9.3	5.5	9.2
	July	224.7	5.4	185	CT	-21.3	-56.1	-43.1	-43.1	-49.3	-84.1
					NT	-16.0	-50.8	-15.6	-50.4	-13.5	-48.3
	August	155.1	0.2	266	CT	0.3	110.9	1.2	111.8	0.9	111.6
					NT	-7.0	103.6	-10.9	99.7	-12.4	98.2
	September	49.1	13	95	CT	-16.2	42.2 a	-25.3	33.2	-29.4	29.0
					NT	-56.9	1.5 b	-109.9	-51.5	-115.2	-56.7
2017	June	47.6	5.2	27	CT	-15.5	-30.9	-20.0	-35.4	-21.6	-36.9
					NT	-5.8	-21.1	-5.4	-20.7	-5.0	-20.4
	July	225.7	33.2	158	CT	-13.9	-48.8	-23.6	-58.5	-21.4	-56.3
					NT	-11.0	-46.0	-12.7	-47.7	-11.5	-46.5
	August	163.7	13.6	95	CT	-36.0	-91.5	-53.0	-108.6	-61.5	-117.1
					NT	-35.9	-91.5	-48.5	-104.1	-53.3	-108.8

Different lowercases indicate statistical differences among tillage system in a given month ($P \leq 0.05$).

ΔW 50, ΔW 100, ΔW 150: are the mean changes in the soil water storage at 50, 100 and 150 cm soil depth; D50, D100, D150: are the cumulative amount of the water drained through the soil profile at 50, 100 and 150 cm.

The results showed in Table 3.4 correspond to the different parameters that were considered in the water balance estimation. As it can be observed in 2015, neither ΔW nor the amount of the drained water showed significant differences among tillage systems in all the considered depths of the study. Although, the mean amount of drained water during July, August and September was not statistically significant between tillage systems, it was 10.9, 12.9 and 14.1% higher in the tilled plots than in the non-tilled ones at 50, 100 and 150 cm soil depths, respectively.

As in the first year of the study, ΔW and the drained water in 2016 (Table 3.4) did not show significant differences among tillage systems in the three studied depths. However, ΔW was higher under NT system than CT during the month of June at the different depths of the soil profile. The mean drained water in August and September was 45.6 % higher in the tilled soil than the undisturbed soil at 50 cm soil depth while no significant difference was observed between tillage system at 100 and 150 cm soil depth in August. Nevertheless, the results obtained in September showed the absence of drained water under NT system at the same depths listed above, compared to an elevated rate of water percolation under CT system.

In the last year of the experiment, the mean ΔW displayed the lowest values and no water percolation was observed (Table 3.4) during the different months of the study. However, no significant differences were observed between tillage systems.

3.3.3. Grain yield components

Table 3.5. Maize yield and yield components according to conventional tillage (CT) and no-tillage (NT) during the 3-year study.

Year	Tillage system	Grain yield (t ha^{-1})	Grain number per ear	1000 grain weight (g)	Ear number per plant	Grain number per row	Row number per ear
2015	CT	17.4 a	490.6 b	599.5 a	1.0 a	27.0 b	18.3 a
	NT	17.8 a	579.0 a	485.9 a	1.0 a	33.0 a	17.5 a
2016	CT	14.4 a	492.4 b	276.2 a	1.1 a	30.3 a	16.1 a
	NT	14.6 a	586.0 a	325.2 a	1.1 a	34.2 a	17.1 a
2017	CT	10.0 a	440.0 b	340.0 a	1.0 b	27.8 a	15.8 a
	NT	10.9 a	517.5 a	325.9 a	1.3 a	29.6 a	17.4 a
Mean	2015	17.6 a	534.8 a	542.7 a	1.0 a	30.0 ab	17.9 a
	2016	14.5 b	539.2 a	300.7 b	1.1 a	32.2 a	16.6 b
	2017	10.5 c	478.8 b	332.9 b	1.2 a	28.7 b	16.6 b
Mean	CT	13.9 a	474.3 b	405.2 a	1.0 b	28.4 b	16.8 a
	NT	14.4 a	560.8 a	379.0 a	1.1 a	32.3 a	17.3 a
Year x Tillage		ns	ns	*	**	ns	*

ns, no significant; * significant at $p < 0.05$; ** significant at $p < 0.01$.
Values followed by the same letter in a row are not significantly different ($p < 0.05$).
Values followed by different letter in a column are significantly different ($p < 0.05$).

Table 3.5 shows the mean effect of tillage systems and years on maize yield and its components. The results indicated that grain yield (estimated using the yield components) did not show significant differences according to tillage systems. However, the main grain yield obtained in 2015 was 21.4 and 67.6 % higher than those obtained in 2016 and 2017, respectively.

Data in Table 3.5 show that the 1000 grain per ear, the ear number per plant and the grain number per row were 18, 10 and 14 % respectively higher under NT system than CT. Although no significant differences were observed, the mean grain yield and row number per ear were 4 and 3 % respectively higher under NT management than CT. However, the 1000 grain weight was 18 % higher in the tilled plots than under NT system without underlining a significant difference.

The results in Table 3.5 indicate that grain number per ear was 10.5 and 12.6% significantly lower in 2017 than in 2015 and 2016 respectively. While, the 1000 grain weight was 80.5 and 63.0 % significantly higher in 2015 than in 2016 and 2017 respectively, and the row number per ear was also 8 % higher in 2015 than both 2016 and 2017. Ear number per plant did not display significant difference among years, although it was 20 and 9 % higher in 2017 than in 2015 and 2016 respectively. The statistical analysis, showed that mean grain yield, grain number per ear and grain number per row were not affected by the interaction year x tillage, but only by the tillage system in one hand and by the year on the other hand.

Maize grain yield had strong and linear response to seasonal ETc and irrigation with high coefficients of determination ($R^2 \geq 0.77$ for ETc and ≥ 0.81 for irrigation) at 100 cm depth under both tillage systems (Figure 3.2). The same coefficients were also obtained for the regression made at 50 and 150 cm depth

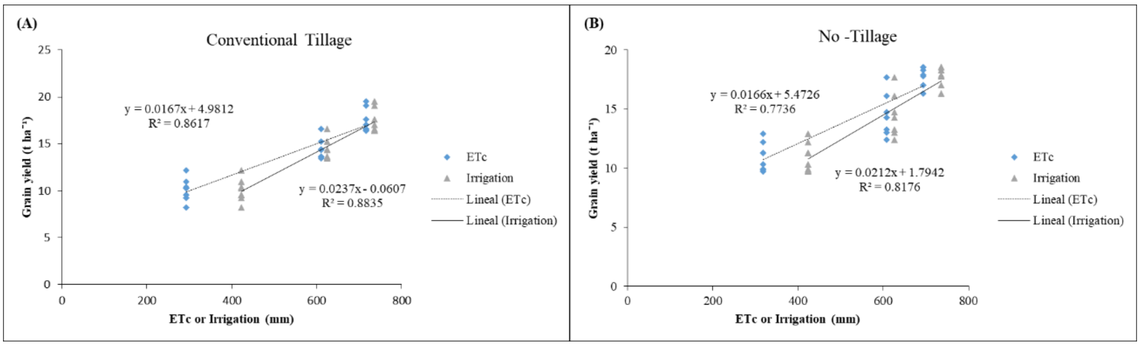

Figure 3.2: Relationship between both seasonal irrigation and evapotranspiration (ETc) and maize grain yield under conventional tillage (A) and no-tillage (B) system.

3.3.4. Water and Irrigation Use efficiency

The estimated WUE and IWUE for both tillage systems in each year are presented in Table 3.6. The WUE did not show significant differences between tillage systems during the 3-year study. It ranged from 2.2 to 2.8 kg m^{-3} under CT system and from 2.3 to 3.0 kg m^{-3} under NT treatment. Depending on the study year, the WUE obtained in 2017 was 16.8 and 20.20% higher than the ones obtained in 2015 and 2016 respectively.

The IWUE did not display significant differences between tillage systems (Table 3.6) during the 3-year study. It ranged from 2.2 to 2.7 kg m^{-3} under CT system and from 2.3 to 2.9 kg m^{-3} under NT treatment. However, the IWUE obtained in 2017 was 11.2 and 23.5% higher than in 2015 and 2016 respectively. These results suggest that IWUE is negatively related to irrigation (R^2= -0.45* and -0.62** under CT and NT respectively).

Table 3.6: Mean water use efficiency (WUE) and irrigation water use efficiency (IWUE) in the first 100 cm depth under conventional tillage (CT) and no-tillage (NT) during the 3-year study.

	WUE (kg m^{-3})		IWUE (kg m^{-3})	
Tillage system	CT	NT	CT	NT
Year				
2015	2.47 a	2.53 a	2.47 a	2.52 a
2016	2.24 a	2.28 a	2.23 a	2.27 a
2017	2.82 a	3.02 a	2.66 a	2.90 a
Mean	WUE (kg m^{-3})		IWUE (kg m^{-3})	
2015	2.50 b		2.50 b	
2016	2.26 c		2.25 c	
2017	2.92 a		2.78 a	

Values followed by the same letter in a row are not significantly different (p< 0.05).
Values followed by different letter in a column are significantly different (p< 0.05).

3.4. Discussion

3.4.1. Soil water dynamics

The SWC presented a general decrease through the soil profile under CT system in 2015 (Figure 3.1. I. a), especially from 60 to 100 cm, but went back to higher values than the PWP at 110cm and below. The low SWC at this interval could have been caused by (1) an irregularity of the irrigation treatments or (2) the tillage did not have effect at these depths and the soil was subject to compaction which prevented the water to flow through the pores. The last irrigation treatment happened on the 8th of June followed by a recorded amount of 7.6 mm of precipitation on the 22nd of June while the ETc reported for this period was higher than the amount of water provided which led to the decrease of the SWC in the soil first 50 cm recorded on the 26th of June 2015. In 2016 (Figure 3.1. II. a; b), the SWC presented a variation that depended mainly on the irrigation treatments and the measurement dates under CT system while NT management presented higher SWC in the first 50 cm of the soil profile. The same variation was also observed in 2017 (Figure 3.1. III. a and b), the SWC depended on the amount of water provided and the measurement dates under CT system while NT management recorded a decrease of the SWC at the lower soil layers (110-150 cm) but presented higher content than CT system in the soil first 40 cm. The last two measurements dates in 2017 (August, 28 and September, 1st) presented the lowest SWC during this season which was caused by the cut-off of irrigation on the second week of August.

The SWC decrease witnessed in 2016 and 2017 in the lower soil profile under NT system is mainly due to the absence of soil disturbance which promotes the moisture accumulation in the first layers (up to 50 cm) while tillage promotes water drainage to lower layers by disrupting soil aggregates explain why CT system showed higher moisture content than NT practice.

During the experiment, the CSWC did not display significant differences according to the reproduction stages of the crop, this could be explained by the continuity of the irrigation treatments all along these stages as a response to the high Etc and mean temperature during summer months (Table 3.2). In 2015, NT system displayed significantly higher CSWC than CT management (Table 3.3), it is necessary to mention that this year was characterised by the highest amount of water provided to the crop all along the production season which could be the reason of this difference. Moreover, the non-disturbance of the soil played a major role in keeping the soil moisture for longer periods than CT system which promotes the deep percolation of water. In both 2016 and 2017, NT system displayed significantly higher CSWC in the soil first 50 cm (Table 3.3), this also highlighted the importance of the non-disturbance of the soil surface combined with the presence of crop residues. Actually, under NT management crop residues are left on the soil surface and are slowly decomposed which helps reducing the evaporation and holds the soil moisture for longer period than CT system when air temperature is a stress factor for the plant. Besides, these residues prevent the loss of soil by erosion when exposed to wind (Shelton et al., 2000). The results obtained in our study are in accordance with the findings of Kenney et al. (2015) who studied the effect of different rates of Stover removal on the soil water content in a maize crop established in two irrigated sites managed under no-till and a third rain-fed region of Kansas managed under strip tillage. The results obtained highlighted the fact that by removing the crop residues from the soil surface SWC generally decrease. In early summer, Stover removal rates of 50 and 100% reduced SWC by about 0.07 m^3 m^{-3} compared to no removal in the top 5 cm. The enhance of soil infiltration under NT system joined to a lower evapotranspiration rate of soil covered by crop residues, conducts many times to greater SWC (Martens, 2000; Nielsen et al., 2005).

However, CT system presented significantly higher CSWC in 2017 (Table 3.3) at 150 cm soil depth than NT system. The soil inversion under CT leads to a faster percolation and a better infiltration of the water situated in the soil surface. Nevertheless, the non-disturbance of the soil and the fact of keeping crop residues in no-tillage plots helped the water to concentrate in the upper layers of the soil thus making it easily available for the plant to use. Also this explains the fact of the decreasing of the available water at 150 cm depth. The results obtained in this study (Table 3.3) underline the fact that NT management could present a viable solution when water is a limiting factor for maize production. These results are in accordance with the results reported by Alvarez and Steinbach (2009) who carried out a meta-analysis of data collected from 35 field experiments with different crops to study the effect of three tillage systems consisting of plough tillage, reduced tillage and no-till management, on the SWC in the Argentine Pampas. They found that NT had greater water content with an average of 16 mm higher than the other tillage systems, while in areas characterized by humid climate SWC was 9 mm greater under NT than under plough and reduced tillage. In semiarid regions and in coarse textured soils, the difference of water content increased to reach 18 mm under NT, which is in agreement with our findings. They also highlighted that NT had an average of 13-14% more water than ploughed soil at both seeding and flowering stages, with a difference of 19 mm in maize. According to Doorenbos and Pruitt (1977) and Alvarez and Steinbach (2009), the water layers available under NT can cover a loss by evapotranspiration ranging from 1 to 3 days of the crop during flowering stage, especially in dry areas with high atmospheric demands. Król et al. (2018) also confirmed, in a long-term experiment on maize crop established in central Poland where the average rainfall is 436 mm that NT system increased SWC compared to CT system. Up to 60 cm depth of a loamy sand soil, SWC under NT system was greater than CT system from 6% in

2014 during the stage of emergence - flowering to 82% in 2016 during the stage of milk - physiological maturity. In Ziway where the rainfall amount ranged from 518 to 1002 mm, Sime et al. (2015) obtained higher soil moisture capturing capacity under zero tillage compared to CT at the flowering and the physiological maturity of maize. They also confirmed the importance of mulching which was able to improve soil moisture content more than no mulch at planting, flowering and the physiological maturity of the crop.

According to Ke et al. (2007), the positive values of the drained water obtained in this study (Table 3.4) mean that deep percolation at the plant root zone or below is occurring and at the different layers of soil. The highest amount of drained water in the three studied soil depths was observed in August and September of 2015 and 2016, and occurred because the water inputs exceeded ETc (Rhoades and Bennett, 1990). However, the negative values that can be observed in Table 3.4 of the water drainage correspond to the capillary rise phenomenon (Ke et al., 2007). This low amount of percolated drained water or its absence through the soil profile is due to the fact that at these months the water inputs did not cover the losses by evapotranspiration and that in 2017, the cut-off of irrigation, the lack of precipitation and the high air temperatures played a major role in promoting the capillary rise phenomenon.

The results variation demonstrates the necessity of the phenological control of the crop since the maize plant has absorbed all the water available at the topsoil and at the root zone during the critical moments of the growing season. Moreover, this control helps identifying the point when it is suitable to cease irrigation treatments to prevent from chemical products from leaching to the aquifers and reduce the economic cost of maize production. The water drainage results obtained in this study (Table 3.4) fall in the range of the findings of Moreno et al. (1996) who reported values that ranged from 142 to 241 in 1992 and from 57 to 85 mm in 1993 mm in a maize crop cultivated on a bare sandy loam (Typic Xerochrept) soil. Generally, CT system promotes the disturbance and the destruction of the soil structure, which leads to water loss throughout the profile. This excess water may be polluted by agrochemicals and soluble nutrients (Fares and Alva, 2000). In addition of the agricultural operations performed on the soil, the texture of this last plays an important role in the infiltration of water into the aquifer.

3.4.2. Grain yield components

According to Fageria et al. (2006) yield components have a direct effect on final maize yield and include the grains per ear and number of ears per plant (or ears per square meter). Moreover, yield components that are considered on a secondary scale are those that indirectly affect yield through their effect on primary components and include rows per ear and grains per row. The results obtained in our study demonstrate that the 1000 grain weight had the strongest effect on maize yield (R^2 = 0.65**) followed by the grain number per ear (R^2 = 0.33*). This was in accordance with the study of Karasu et al. (2015) who reported that grain yield depended on the 1000 rain weight (R^2 = 0.94**) and grain number per ear (R^2 = 0.86**). While the results obtained by Greaves and Wang (2017a) showed that maize grain yield was mostly influenced by the grain number per ears (R^2 = 0.90**) then by the 1000 grain weight (R^2 = 0.87**). In all cases, these two parameters influenced directly the variation of maize grain yield. The ear number per plant has the weakest effect on grain yield (R^2 = 0.14), as it can be observed in Table 3.5, the last year of the study (2017) has the highest number of ears per plant under NT system and yet the grain yield obtained was the lowest. The lack of available water during the grain filling stages of the plant (Table 3.3) led to a drastic drop of grain yield in 2017 compared to the previous years. Moreover, the decrease of the 1000 grain weight in 2016, could be caused by the lack of water during the first reproductive stages in July (Table 3.4) which are considered the most critical stages according to (Rhoades and Bennett, 1990) and are the ones determining the grain yield at harvest. This could be confirmed by the fact that in 2015, the 1000 grain weight was the highest which could be related to the fact that during these same stages, the crop has received an amount of water that covered the losses by evapotranspiration and even led to a water loss by percolation (Table 3.4).

The linear relationship observed suggests that yield would decrease as either ETc or irrigation treatments are reduced. The results obtained in our study are in accordance with the ones found by Irmak et al. (2019) and Greaves et al. (2017 b) who reported strong relationship between maize grain yield, ETc and irrigation. While Payero et al. (2008) obtained polynomial relationship in the case of yield versus irrigation and where they stated that excessive irrigation tends to decrease maize grain yield. A study conducted by Greaves and Wang (2017a) demonstrated differences in maize grain yield in Taiwan when they applied different irrigation treatments: the higher yields were obtained in treatments where 100% of water was available while they decreased significantly in treatments with water deficit. They recorded a maize yield of 10.1 t ha^{-1} under fully irrigation (I_1 = 60 mm) in comparison with a quantity of 9.4, 9.3, 7.7 and 6.8 t ha^{-1} under 83, 67, 50 and 33 % of I_1, respectively. The same variation was also observed in Karasu et al. (2015) works' and who reported an increase of the maize grain yield from17.8 to 18.3 t ha^{-1} when they increased the irrigation amount from 826 to 1027 mm. Payero et al. (2008) reported also higher mean maize yield (9.87 t ha^{-1}) under an average amount of irrigation of 291 mm in comparison with an average yield of 6.50 t h^{-1} under a treatment of 37.5 mm.

Although grain yields are not statistically different during our 3-year study, NT presented numerically higher yield than CT thanks to a higher SWC in the first 100 cm. These results are supported by Munodawafa and Zhou (2008) where they recorded higher maize yield under mulch ripping (MR) and tied ridging (TR) than under conventional tillage in a study based only on rainfall. The higher yields recorded under MR and TR during the drier seasons highlight the moisture conservation potential of these treatments. This would mean that even during years when rainfall is low and unreliable, good yields (although low) can still be attained under conservation tillage unlike under CT (Rockstrom, 2003). Copec et al. (2015) recorded on the first year of their study, which witnessed a lack of precipitation (653.4 mm) and low SWC, lower maize yields ranging from 5.4 t ha^{-1} under chisel plough and multitiller (CM) to 5.1 t ha^{-1} under CT and 4.5 t ha^{-1} under NT. However, in the following years, the highest yields were achieved under CM and NT systems, under which the highest SWC was measured while the lowest yields were achieved under CT, under which the lowest SWC was also measured. Copec et al. (2015) found a strong correlation (R^2 = 0.72 **) between yield and all tillage system. In addition, Lamm et al. (2009) reported an average maize yield of 13.4 t ha^{-1} under NT treatments while plough-based treatments recorded a mean of 12.2 t ha^{-1} and concluded that greater irrigation capacity generally increased grain yield in a four-year study. In a long-term tillage experiment (1980-1987) of two seasons, Lal (1997) reported mean grain yield of 3.1, 3.5 and 3.0 t ha^{-1} under NT without mulching, NT with mulching and ploughing treatments, respectively for an amount of precipitation of an average of 687+113 mm. In addition, a grain yield of 1.3, 1.5 and 1.5 t ha^{-1} for the same treatments respectively for an average amount of 524+11mm. In researches on wheat yield response, Hemmat and Eskandari (2004) found that under conservation tillage practice; grain yield is variable but higher ones are usually attributed to increased water conservation.

3.4.3. Water and Irrigation Use efficiency

The WUE displayed the highest value in 2017 (Table 3.6) in comparison with both 2015 and 2016, this could mean that, on average, when the amount of available water is low, less grain yield is produced per mm of water. In this study, the WUE was the highest in 2017 but ETc (356.2 and 362.2 mm under CT and NT respectively) during the maize reproductive stages and the grain yield were the lowest. In contrast, the high-irrigated year (2015) displayed high grain yield and ETc (706.2 and 702.8 mm under both tillage systems but moderate WUE, while 2016 had a decent grain yield and ETc (639 mm under CT and NT respectively) but the lowest WUE. The increase of WUE in 2015 compared to 2016 could be attributed to the noticeable difference in grain yield between the two years. It could also be linked to an increased leaf area and its effects on the soil evaporation to crop transpiration (Zhang et al., 1998). Karam et al. (2003) suggested that the increase in the WUE in water stressed plants is the result of a larger decline in plant transpiration due to reduced leaf area as a consequence of water deficit. Generally, the results obtained in our study could confirm that WUE is negatively related to irrigation level and ETc (R^2= -0.64** and -0.70** under CT and NT system respectively). These findings agree with the results obtained by Kresović et al. (2016) in Vojvodina region, where they recorded a lower WUE for a full-irrigated maize (2.8 kg m^{-3}) in comparison with deficit irrigation treatment where WUE ranged from 3.1 to 3.3 kg m^{-3}. Zhang et al. (2017) reported higher WUE for an amount of irrigation of 420 mm than in a full-irrigation (600mm) in three high yield maize hybrids and were ranged from 2.61 kg m^{-3} to 2.9 kg m^{-3} with deficit irrigation treatment and from 2.3 to 2.7 kg m^{-3} without applying any water stress to the plant. Garcia y Garcia et al. (2009) found that rainfed sweet maize presented higher WUE with 2.7 kg m^{-3} in comparison with the irrigated one that displayed a value of 2.0 kg m^{-3}. However, Greaves and Wang (2017a) recorded lower values, ranging from 1.5 to 2.3 kg m-3 but with the same trend as the ones recorded in our study where the higher WUE corresponded to deficit irrigation and the lower to a full irrigation treatment.

In this study, the ranges of IWUE fall in the interval reported by Greaves and Wang (2017a) going from 1.6 to 4.5 kg m^{-3} and being the lowest and the highest with full and deficit irrigation respectively. Di Paolo and Rinaldi (2008) also reported ranges of IWUE going from 2.5 to 3.0 kg m^{-3}, which are close to our findings and confirmed the negative relationship between IWUE and irrigation because when they applied 50% ETc, IWUE reached 3.3 kg m^{-3} while 100% ETc led to an IWUE of 2.3 kg m^{-3}. In addition, Payero et al (2008) confirmed these results, when they highlighted the decreasing of IWUE values by increasing the amount of water application. They achieved IWUE values of 15.9 and 21.1 kg m^{-3} for water applications of 53 and 22 mm, in comparison with values of 2.9 and 4.1 kg m^{-3} for water applications of 356 and 226 kg m^{-3} in 2005 and 2006, respectively. In contrast to our results, Farré and Faci (2006) found that by reducing the irrigation amount in maize crop in Northeast Spain, the IWUE decreases. Kresović et al. (2016) also declared that by applying higher irrigation amount IWUE values are maximized and ranged from 3.5 to 3.6 kg m^{-3} when full irrigation is applied and from 1.1 to 2.9 kg m^{-3} with a lower amount.

Although tillage system did not show significant differences, the WUE and IWUE were numerically higher under NT system than CT treatment (Table 3.6). This could be explained by a lower ETc in NT plots thanks to the presence of crop residues on the surface soil, which form a barrier to evaporation from soil and help holding more moisture in the topsoil in addition of a numerically higher grain yield. In this case, NT system could be used as an alternative to increase WUE and lower IWUE (which is the case in Table 3.6) if the water is the limiting factor for maize production. Moreover, the variation from year to year depends on the climatic conditions of the region and the maize variety planted. For instance, the grain yield obtained in 2016 (Table 3.5) is considered low for the Roxxy variety that displayed higher grain yield in 2015. A 91.5 % reduction in the irrigation amount from 2015 to 2016 resulted in a reduction of 82% in grain yield under both tillage systems. This reduction would affect drastically the financial balance therefore; farmers should maximize the economic return per unit water used than per land unit (Zhang et al., 2017).

3.5. Conclusion

This study highlights the variation of the SWC under CT and NT managements. The first year (2015) of the trial consisted of a good example to confirm this significant difference, where NT was the treatment with higher CSWC at the different soil depths and led to numerically higher grain yield than CT system. The second year (2016) displayed only a significant difference at the first 50 cm soil layers where CSWC under NT was higher than CT treatment, this may also be the reason why at the end of the season the higher yield was obtained under NT system. Even though, this difference was not significant in 2017, NT system displayed numerically higher yield throughout the drought period during the maize growth season. This was thanks to a higher SWC in the first 100 cm of the soil. The results obtained also underlined an excess of water, which translated in drainage at the root zone of the plant and below in 2015 and 2016 and the lack of it resulted in a capillary rise in 2017. This work, also underlined the importance of the amount of water applied to the crop, the assessment of water productivity helped find that WUE and IWUE were higher in 2017, where the lowest amount of water was applied. It also brought out the fact that by increasing irrigation amount WUE and IWUE values decreased. Therefore, if the water is the limiting factor for maize production, which is the case in Castile and Leon, increasing WUE could be one of the solutions to limit the water waste without penalizing grain yield and the economic benefit. The water productivity in our study did not show significant differences between tillage treatments.

In this case, managing maize irrigation at the field scale can be improved by quantifying the water balance and using advanced techniques for irrigation scheduling for more effective and economic use of limited water supplies. Deficit or regulated deficit irrigation is one strategy for maximizing the water use efficiency for higher yield per unit of irrigation, and it should be considered as a solution to reduce the number of irrigation events without causing a drastic decrease of grain yield. However, judicious planning is required so that water stress is minimized in the critical growth stages of the crop. It is also necessary to mention the importance of crop residues left on the soil in the decreasing of evaporation, the run-off and temperature. Moreover, a good selection of the crop genotype that adapts to the climatic conditions is an essential factor to ameliorate the WUE and/or IWUE.

References

Allen, R.G., Pereira, L.S., Raes, D. Smith, M. 1998. Crop Evapotranspiration: Guidelines for computing crop water requirements. FAO Irrigation and Drainage Paper No.56; 300p.

Alvarez, R., Steinbach, H.S. 2009. A review of the effects of tillage systems on some soil physical properties, water content, nitrate availability and crops yield in the Argentine Pampas. Soil & Tillage Research, 104: 1-15.

Arshad, M.A., Franzluebbers, A. J., Azooz, R.H. 1999. Components of surface soil structure under conventional and no tillage in north-western Canada. Soil & Tillage Research, 53 : 41-47.

Chopart, J.L., Kone, D. 1985. Influence de differentes techniques de travail du sol sur l'alimentation hydrique du maïs et du cotonnier en Côte d'Ivoire. Agronomie Tropicale. 40: 223-229.

Copec, K., Filipovic, D., Kovacev, I., Kosutic, S. 2015. Effects of tillage systems on soil water content and yield in maize and winter wheat production. Plant Soil Environment. Vol 61, no. 5, 213-219.

Derpsch, R. 2001. Conservation tillage, no-tillage and related technologies, in Proc. 1st World Conserv. Agric., Madrid, October 1-5, 2001Conservation Agriculture. A Worldwide Challenge, Vol. I, 161-170.

De Vita, P., Di Paolo, E., Fecondo, G. Di Fonzo, N., Pisante, M. 2007. No-tillage and conventional tillage effects on durum wheat yield, grain quality and soil moisture content in southern Italy. Soil & Tillage Research, 92: 69-78.

Di Paolo, E., Rinaldi, M. 2008. Yields response of corn to irrigation and nitrogen fertilization in a Mediterranean environment. Field Crops Research, 105: 202-210.

Doorenbos, J., Pruitt, W.O. 1997. Crop water requirements. FAO. Irrigation and Drainage Paper No. 24, Rome, Italy, p.139.

Fares, A., Alva, A. 2000. Soil Water Components Based on Capacitance Probe in a Sandy Soil. Soil Science Scociety of America Journal. 64, 311- 318.

Farré, I., Faci, J.M. 2006. Comparative response of maize (*Zea mays* L.) and sorghum (*Sorghum bicolor* L. Moench) to deficit irrigation in a Mediterranean environment. Agricultural Water Management, 83: 135-143.

FAO 2004. Kerry turner, Stavors Georgiou, Rebecca Clark, Roy Brouwer and Jacob Burke. Economic evaluation of water in Agriculture, from the sectoral to a functional perspective of natural resource management.

FAO 2017. Coping with water scarcity in agriculture, a global framework for action in a changing climate. Rome: FAO.

Garcia y Garcia, A., Guerra, L.C., Hoogenboom, G. 2009. Water use and water use efficiency of sweet corn under different weather condictions and soil moisture regimes. Agricultural Water Management, 96: 1369-1376.

Greaves, G.E., Wang, Y. M. 2017a. Yield response, water productivity and seasonal water production functions for maize under deficit irrigation water management in southern Taiwan. Plant Production. Science. Vol 20, no 4, 1-13.

Greaves, G.E., Wang, Y. M. 2017b. Effect of regulated irrigation scheduling on water use of corn in southern Taiwan tropical environment. Agricultural. Water Management, 188: 115-125.

Groves, S.J., Rose, S.C. 2004. Calibration equations for Diviner 2000 capacitance measurements of volumetric soil water content of six soils. Soil Use and Management, 20: 96-97.

Guzha, A. C. 2004. Effects of tillage on soil microrelief, surface depression storage and soil water storage. Soil & Tillage Research, 76: 105-114.

Haberland, J. Gálvez, R. Kremer, C. Zuñiga C. Rudolffi. Y. 2015. Accurate soil water content monitoring in real time with appropriate field calibration of the FDR device “Diviner 2000” in a commercial table grape vineyard. International Journal of Science and Technology, Vol 4, no 2, 273-284.

Hemmat, A., Eskandari, I. 2004. Conservation tillage practices for winter wheat-fallow farming in the temperate continental climate of northwestern Iran. Field Crops Research, 89: 123-133.

Heisey, P.W., Edmeades, G. O. 1999. Maize production in drought-stressed environments: technical options and research resource allocation. World Maize Facts and Trends 1997/1998.

Kang, S.Z., Hao, X.M., Du, T.S., Tong, L., Su, X.L., Lu, H.N., Li, X.L., Huo, Z.L., Li, S.E., Ding, R.S. 2017. Improving agricultural water productivity to ensure food security in China under changing environment: from research to practice. Agricultural Water Management, 179: 5-17.

Karam, F., Breidy, J., Stephann, C., Rouphael, J. 2003. Evapotranspiration, yield and water use efficiency of drip irrigated corn in the Bekaa Valley of Lebanon. Agricultural Water Management 63, 125-137.

Karasu, A., Kuscu, H., Öz, M., Bayram, G. 2015. The effect of different irrigation water levels on grain yield, yield components and some quality parameters of silage maize (Zea mays indentta Sturt) in Marmara region of Turkey. Notulae Botanicae Horti Agrobotanici Cluj-Napoca, 43, 138-145.

Ke, J., Cornelis, W.M., Schiettecatte, W., Yao, Y., Wu, H., Gabriels, D., De Neve, S., Cai, D., Jin, J., Hartmann, R. 2007. Effects of different management practices on the soil-water balance and crop yield for improved dryland farming in the Chinese Loess Plateau. Soil & Tillage Research. 96, 131-144.

Kenney, I., Blanco-Canqui, H., Presley, D.R., Rice, C.W., Janssen, K., Olson, B. 2015. Soil and crop response to stover removal from rainfed and irrigated corn. Glob. Change Biol. Bioenergy 7: 219-230.

Kijne, J.W., Baker, R., Molden. D.J., 2003. Water Productivity in Agriculture: Limits and Opportunities for Improvement. CABI.

Kresović, B., Tapanarova, A., Tomić, Z., Životić, L., Vujović, D., Sredojević, Z., Gajić, B. 2016. Grain yield and water use efficiency of maize as influenced by different irrigation regimes through sprinkler irrigation under temperate climate. Agricultural Water Management, 169: 34-43.

Król, A., Żyłowski, T., Kozyra, J., Księżak, J. 2018. Soil moisture under no-tillage and tillage systems in maize long-term experiment. Polish J. Soil Sci. Vol. LI/1, 104-117.

Kuscu, H., Karasu, A., Öz, M., Demir, A. O., Turgut, I. 2013. Effect of irrigation amounts applied with drip irrigation on maize evapotranspiration, yield, water use efficiency and net return in a sub-humid climate. Turkish Journal Field Crops, 18: 13-19.

Lal, R., Maurya, P.R., Osei-Yeboah, S. 1978. Effects of no-tillage and plowing on efficiency of water use in maize and cowpea. Experimental Agriculture, 14: 113-120.

Lal, R. 1991. Current research on crop water balance and implications for the future. Soil Water Balance in the Sudano-Sahelian Zone (Proceedings of the Niamey Workshop, February 1991). IAHS. Publ. no. 199, 1991.

Lal, R. 1997. Long-term tillage and maize monoculture effects on a tropical Alfisol in western Nigeria. I. Crop yield and soil physical properties. Soil & Tillage Research, 42: 145-160.

Lamm, F.R., Aiken, R.M., Abou Kheira, A. A. 2009. Corn yield and water characteristics as affected by tillage, plant density, and irrigation. Transactions of the ASABE. Vol. 52 (1), 133-143.

Lampurlanés, J., Angás, P., Cantero-Martínez, C. 2001. Root growth, soil water content and yield of barley under different tillage on two soils in semiarid conditions. Field Crops Research, 69: 27-40.

Liu, Z.D., Xiao, J.F., Liu, Z.G., Nan, J.Q., Yang, D.C. 2011. Effects of different irrigation treatment on morphological indexes, water consumption and yield maize under mulch drip irrigation. Journal of Irrigation and Drainage Engineering, 30: 60-40.

Martens, D.A. 2000. Nitrogen cycling under different soil management systems. Advances in Agronomy, 70: 143-191.

McMaster, G.S., Palic, D. B., Dunn, G.H. 2002. Soil management alters seedling emergence and subsequent autumn growth and yield in dryland winter wheat- fallow systems in the Central Great Plains on a clay loam soil. Soil & Tillage Research, 65: 193-206.

Moreno, F., Cayuela, J.A., Fernández-Boy, E., Murillo, J.M., Cabarera, F. 1996. Water balance and nitrate leaching in an irrigated maize crop in SW Spain. Agricultural Water Management, 32: 71-83.

Munodawafa, A. Zhou, N. 2008. Improving water utilization in maize production through conservation tillage systems in semi-arid Zimbabwe. Phys. Chem. Earth. 33, 757-761.

Nielsen, D.C., Unger, P., Miller, P.R., 2015. Efficient water use in dryland cropping systems in the Great Plains. Agronomy Journal, 97: 364-372.

Paraskevas, C., Georgiou, P., Ilias, A., Babajimopoulos. 2012. Calibration for two capacitance water content probes. Agrophys, 26: 285-293.

Payero, J.O., Tarkalson, D.D., Irmak, S., Davison, D., Petersen, J.L. 2008. Effect of irrigation amounts applied with subsurface drip irrigation on corn evapotranspiration, yield, water use efficiency, and dry matter production in a semiarid climate. Agricultural Water Management, 95: 895-908.

Rasmussen, K.J. 1999. Impact of plough less soil tillage on yield and soil quality: A Scandinaviam review. Soil & Tillage Research, 53: 3-14.

Reinbott, T.M., Conley, S.P., Blevins, D. G. 2004. Tillage and cropping systems – No-Tillage corn and grain sorghum response to cover crop nitrogen fertilization. Agronomy Journal, 96: 1: 1158-1163.

Richards, L.A. 1941. A Pressure-Membrane Extraction Apparatus for Soil Solution. Soil Science, 51: 377- 386.

Rockstrom, J. 2003. Resilience building and water demand management for drought mitigation. Physics and Chemistry of the Earth, 28: 869-877.

Rhoades, F. M., and Bennett, J. M. 1990. Corn. In: Irrigation of Agricultural Crops (Agronomy Monograph No. 30), B. A. Stewart, and D. R. Nielsen (Eds.). Madison, WI, Am. Soc. Agron. pp. 569–597.

Sentek. Diviner 2000: user guide version 1.21. Stepney: Sentek Pty Ltd, 2000.

Shelton, D.R., Sadeghi, A. M., McCarty, G.W. 2000. Effect of soil water content on denitrification during cover crop decomposition. Soil Science, 165(4): 365-371.

Sime, G., Aune, J.B., Mohammed, H. 2015. Agronomic and economic response of tillage and water conservation management in maize, central rift valley in Ethiopia. Soil & Tillage Research, 148: 20-30.

Stone, L.R., Schlegel, A.J., Khan, A.H. 1996.Response of corn, grain sorghum, and sunflower to irrigatioj in the high Plains of Kansas. Agricultural Water Management, 30: 241- 259.

Stone, P.J., Wilson, D.R., Reid, J.B., Gillespie, R.N. 2001. Water deficit effects on sweet corn. I. Water use, radiation use efficiency, growth, and yield. Australian Journal of Agricultural Research,, 52: 103-11.

Titi, A.E. 2003. Soil Tillage in Agroecosystems. CRC Press.

Topp, G.C., Davis, J.L., Annan, A.P. 1980. Electromagnetic determination of soil-water content: Measurments in coaxial transmission lines. Water Resources Res. 16, 574-582.

Wilhelm, W.W., Schepers, J.S., Mielke L.N., Doran, J.W., Ellis, J.R., Stroup, W.W. 1987. Dryland maize development and yield resulting from tillage and nitrogen fertilization practices. Soil & Tillage Research. 10,167-179.

WMO, 2012 (Agrometeorology of some selected crops, In Guide to Agricultural Meteorological practices (GAMP) (2010 ed., pp.1-128). Geneva: Chair publications Board).

Yazar, A., Howell, T.A., Dusek, D.A., Copeland, K.S. 1999. Evaluation of crop water stress index for LEPA irrigated corn. Irrigation Science, 18 : 171-180.

Zhang, H., Oweis, T.Y., Garabet, S., Pala, M. 1998. Water-use efficiency and transpiration efficiency of wheat under rain-fed conditions and supplemental irrigation in a Mediterranean-type. Plant Soil, 201: 295-305.

Zhang, G., Liu, C., Xiao, C., Xie, R., Ming, B., Hou, P., Liu, G., Xu, Shen, D., Wang, K., Li, S. 2017. Optimizing water use efficiency and economic return of super high yield spring maize under drip irrigation and plastic mulching in arid areas of China. Field Crops Research, 211: 137-146.

CHAPTER 4. SOIL ORGANIC CARBON ACCUMULATION AND CARBON DIOXIDE EMISSIONS IN IRRIGATED CONTINUOUS MAIZE UNDER TWO TILLAGE SYSTEMS IN SEMIARID MEDITERRANEAN CONDITIONS

Dachraoui, M., Sombrero, A. 2021. Soil organic carbon accumulation and carbon dioxide emissions during a 6-year study in irrigated continuous maize under two tillage systems in semiarid Mediterranean conditions.

Spanish Journal of Agricultural Research, Volume 19, Issue 1, e1102.

4.1. Introduction

Maize crop (*Zea mays* L.) is one of the most important crops worldwide and its production is mainly assured by conventional methods as the frequent ploughing of the soil. The use of mouldboard plough for crop residues management and soil preparation is a common practice among farmers. In short-term period, conventional tillage (CT) creates a good soil environment for crop growth, while in a long-term period, this practice would promote the soil erosion and degradation and would increase the mineralization and the depletion of soil organic matter (SOM) (Lenka and Lal, 2013). The conversion from CT to no-tillage (NT) would improve the soil quality and water retention, diversify the soil fauna and reduce the potential for soil erosion and the loss of soil organic carbon (West and Post, 2002; Halvorson et al., 2008). The soil organic carbon (SOC) is a significant indicator of the soil quality as it helps to improve its structure, ameliorate the crop/crop residue ratio and mitigate the effects of climate (Lal, 2007) which is suffering drastic changes caused by the increase of the greenhouse gases concentration (GHGs).

In the agricultural sector, CO_2 is released during the burning of fossil fuels, the use of agricultural machinery, the production of synthetic fertilizers and pesticides, the microbial decomposition and the burning of stubble and SOM (Lal, 2004). However, the land use changes have a double synergistic effect, as a sink (carbon (C) sequestration increase) and as mitigation (reduction of emissions). Soils can function as either a source or a sink for atmospheric GHG depending on land use and soil management. Appropriate management can enable agricultural soils to provide a net sink for sequestering atmospheric CO_2 and other GHG (Paustian et al., 1997a; West and Post, 2002). Agricultural operations such CT promotes the rapid oxidation processes and the release of a large CO_2 amount into the atmosphere, decreasing the levels of OM and contributing to the global warming. Conservation agriculture such as NT practices improves the soil structure, water retention and helps the nutrients preservation. The non-disturbance of the soil and the remaining of the crop residues on the soil surface promoted the increase of the SOC thanks to the reduction of the SOM mineralisation (Balota et al., 2004, Sombrero and De Benito, 2010). The alteration of soil profile increases the flux of CO_2 emissions into the atmosphere and begins immediately after conducting the operation. Therefore, management and land use could be applied to mitigate GHG emissions by sequestring C in the soil and creating a sink for atmospheric CO_2 (Paustian et al., 1997b). No-tillage system could play an important role by increasing SOC and improving the environmental quality in the production systems (Reicosky, 1997) and would be a viable alternative to stabilize CO_2 concentrations in the atmosphere and a way to counteract climate change.

The literature indicated contradictory results respect to tillage effects on CO_2 emissions as Franzluebbers et al. (1995) reported similar or more CO_2 fluxes under a 9 years old NT management compared to CT system while Al-Kaisi and Yin (2005) observed significantly lower CO_2 emission from NT system than CT during the short period after tillage disturbance. Vinten et al. (2002) found higher CO_2 emissions for some periods and lower for others under NT compared to CT. Differences of CO_2 emissions may be the result of short- and long-term effects (Ussiri and Lal, 2009). Pareja-Sanchez et al. (2019) found that in the first and second year of experiment, cumulative CO_2 emissions were greater under NT compared to CT, while in the third year, no differences were found between tillage systems in maize growing season. In 2019, the largest non-tilled lands were recorded in Castile and Leon with an area of 2.492.437 ha (MAPAMA, 2019), nevertheless few studies were conducted and limited information is available on the effect of NT system on SOC and CO_2 emissions in irrigated crops in semiarid areas. To maintain the soil quality, the crop productivity, and to contribute to the mitigation of the GHGs, it is necessary to investigate changes in SOC accumulation and CO_2 emissions and to identify tillage systems that enhance soil conservation. Therefore, this study aims to evaluate the effects of NT and CT managements on the SOC changes, the CO_2 emissions and its relation with both soil temperature and moisture and grain yield in a monoculture of irrigated maize during six years in a semiarid region of Castile and Leon.

4.2. Materials and methods

4.2.1. Site, treatments and experimental design

To avoid repetition, the experimental design and crop management methodology are detailed in ***Chapter 3.***

The climatic conditions are classified as continental Mediterranean and are characterized by cold winter and warm summer with a mean annual temperature of 12.7 °C. The lowest temperatures recorded were in January and the highest in July and August. The annual precipitation reached 405.6 mm and was concentrated from September to May (85%).

The data obtained were collected at the meteorological Zamadueñas station situated at 200 m from the experimental site and are detailed in Table 4.1.

Table 4.1. Mean air temperature and total precipitation (mm) in growing seasons 2011-2017 and historic mean values (1981-2010) at Zamadueñas experimental station, Spain.

	Mean Temperature (ºC)								Total precipitation (mm)							
Months	2011	2012	2013	2014	2015	2016	2017	1981-2010	2011	2012	2013	2014	2015	2016	2017	1981-2010
January	4.6	2.2	4.6	5.9	1.7	6.2	2.5	4.3	46.5	28.4	36.8	22.8	28.0	116.0	10.3	38.2
February	5.1	2.6	4.0	5.4	4.3	5.8	7.0	5.8	29.8	1.0	31.2	48.1	16.4	38.8	39.9	23.9
March	7.9	8.6	7.1	8.7	8.4	6.1	9.2	9.0	44.0	7.6	117.9	11.0	16.8	32.2	6.0	23.3
April	13.7	8.6	9.5	13.0	11.8	9.1	12.6	10.3	49.4	66.8	28.6	22.4	66.0	99.4	3.8	41.9
May	16.4	16.0	11.0	14.1	15.7	13.1	16.7	14.5	37.0	20.2	27.9	18.8	19.8	47.4	42.0	46.0
June	18.3	19.2	16.3	18.5	19.7	19.1	22.4	19.3	18.6	12.6	33.6	9.6	76.2	1.9	5.4	28.7
July	19.7	20.4	23.0	20.3	23.7	23.0	22.4	22.3	0.0	12.2	9.4	66.2	4.2	5.4	33.2	14.0
August	21.0	21.6	20.9	20.6	21.1	22.4	21.8	22.1	34.8	1.4	29.0	0.2	5.2	0.2	13.6	15.0
September	18.5	17.4	17.6	18.5	16.1	18.6	17.5	18.5	0.0	21.8	0.0	61.2	23.6	13.0	0.2	30.2
October	13.1	12.0	13.2	15.0	12.8	13.5	14.7	13.2	17.2	72.6	23.2	37.0	54.2	46.2	7.2	53.9
November	8.5	7.4	6.8	9.3	8.3	6.9	6.1	7.9	64.4	60.4	3.2	71.4	46.8	48.9	17.6	45.7
December	4.0	5.1	2.7	3.8	5.3	4.1	4.0	5.0	11.0	21.4	1.8	17.2	18.4	12.6	26.2	44.1

4.2.2. Soil sampling and analysis

At the outset of the study and during the following years, the SOC was determined at by collecting soil samples after the maize harvest in 2011, 2013, 2015 and 2017 at three sites in each elementary plot to obtain a composite sample per plot at depths of 10, 20 and 30 cm. The samples were air dried and sieved through a 2 mm mesh. The total C and SOC contents were determined by dry combustion with a LECO CNS 1934.

The SOC expressed in Mg ha^{-1} was calculated in terms of elemental soil mass considering the concentration of OC (kg Mg^{-1}), the bulk density, which was similar in both tillage systems (Mg m^{-3}) and depth (m).

Soil CO_2 fluxes were measured with an EGM-4 2000 soil respiration chamber (PP Systems International, Amesbury, MA, USA), which is a manual system composed of an EGM-4 IRGA (InfraRed Gas Analyzer) system linked to a cylindrical soil respiration chamber SRC-1 (diameter 10 cm, height 15 cm). This system makes "Auto-zero" in order to adapt to environmental conditions and afford a stability of the CO_2 signal. The device is a closed dynamic chamber system, which is used to measure the variation of CO_2

exchange with the soil surface during a precise time. The chamber is directly inserted about 1-2 cm deep in the soil surface and the airflow rate was adjusted to 900 ml min^{-1}. Soil CO_2 fluxes were considered as the difference of CO_2 concentration when the air flows through the chamber and when it leaves. After 2 minutes, CO_2 fluxes were recorded and the readings were taken when CO_2 flux was stable enough to prevent from possible unrealistic values that could be caused by the disturbance produced after placing the chamber into the soil (Pumpanen et al., 2004). Measurements were taken twice in every plot in order to corroborate a correct data set. The short-term influence of tillage on soil CO_2 evolution was assessed by recording series of successive measurements during the soil´s preparation and sowing. These measurements were recorded before any field operation took place, then immediately after and at 2, 4, 24 and 48 h after each operation and during maize growing seasons in both CT and NT systems. Annual total soil CO_2 flux was obtained by summing all the measured and interpolated hourly values, total micro-mol of CO_2 for the year was converted into kg CO_2 ha^{-1}. Cumulative CO_2 emissions were quantified on a mass basis (Mg ha^{-1}) using the trapezoid rule.

From 2012 to 2017, soil temperature was measured with a hand-held probe (model STP-1, PPSystems) which was inserted at 5 cm into the soil away from the edge of the CO_2 chamber. A soil temperature value was recorded at the same time as the soil CO_2 flux was recorded. From 2015 to 2017 a soil-surface sample were collected at 10 cm depth along CO_2 measurements to determine the gravimetric soil water content. To determine the biomass and grain yield of the maize crop, plants samples were picked in one-meter area from four rows and were weighed. Furthermore, in every plot, two strips of 12m x 1.5 m were harvested and grains were weighed separately to estimate the crop yield at maturity in October-November.

4.2.3. Statistical analysis

Data were statistically analyzed using the general linear model (GLM) procedure (SAS Institute. 9.4) applying Tukey's test at 5% significant level ($P \leq 0.05$). Regression models were fitted to the date to describe the relationship between climatic variables and CO_2 emissions. Spearman and Pearson correlation was calculated to determine the relationship between soil CO_2 flux and both soil temperatures and moistures.

4.3. Results

4.3.1. Air temperature and precipitation

During the maize cycle, the coldest months were April and November (Table 4.1). Mean temperatures in April (2011, 2014, 2015 and 2017) and November (2011, 2014 and 2015) were warmer than the one recorded in 1981-2010. The third year (2013) recorded the lowest temperatures compared to long-term means. The warmest temperatures in this studied period occurred in July when the mean temperatures were warmer in 2013, 2015 and 2016 than the long-term mean. The warmest year was 2017 followed by 2015. Generally, the months of least precipitation were July and August highlighting that in 2012, 2015 and 2016, the rainfall scarcity combined with high temperatures and evapotranspiration led to the increase of the maize hydric needs. The higher precipitation in June and August 2013 and in July 2014 led to the decrease of the number and the amount of irrigation treatments. During 2017, the combination of the precipitation scarcity and the high temperatures from March to July, led to earlier irrigation treatments that were cut off on August 11, because of the water lack in the region.

4.3.2. Soil temperature and moisture

Soil temperature was recorded from tillage to crop maturity during the six years of the study. The lowest soil temperature occurred in winter and late August-September while the warmest was reported from June to late August (Figure 4.1). Soil temperature reported under NT system was significantly cooler in 2012 (mean difference 1.3 ºC) and 2017 (mean difference 2.3ºC) than under CT. In 2013, 2014 and 2016 low and generally non-significant differences were recorded between both tillage systems (under NT from 0.7 to 0.9ºC lower than CT). Nevertheless, from December to March 2015, soil temperature was slightly warmer (0.9 ºC) under NT than under CT system.

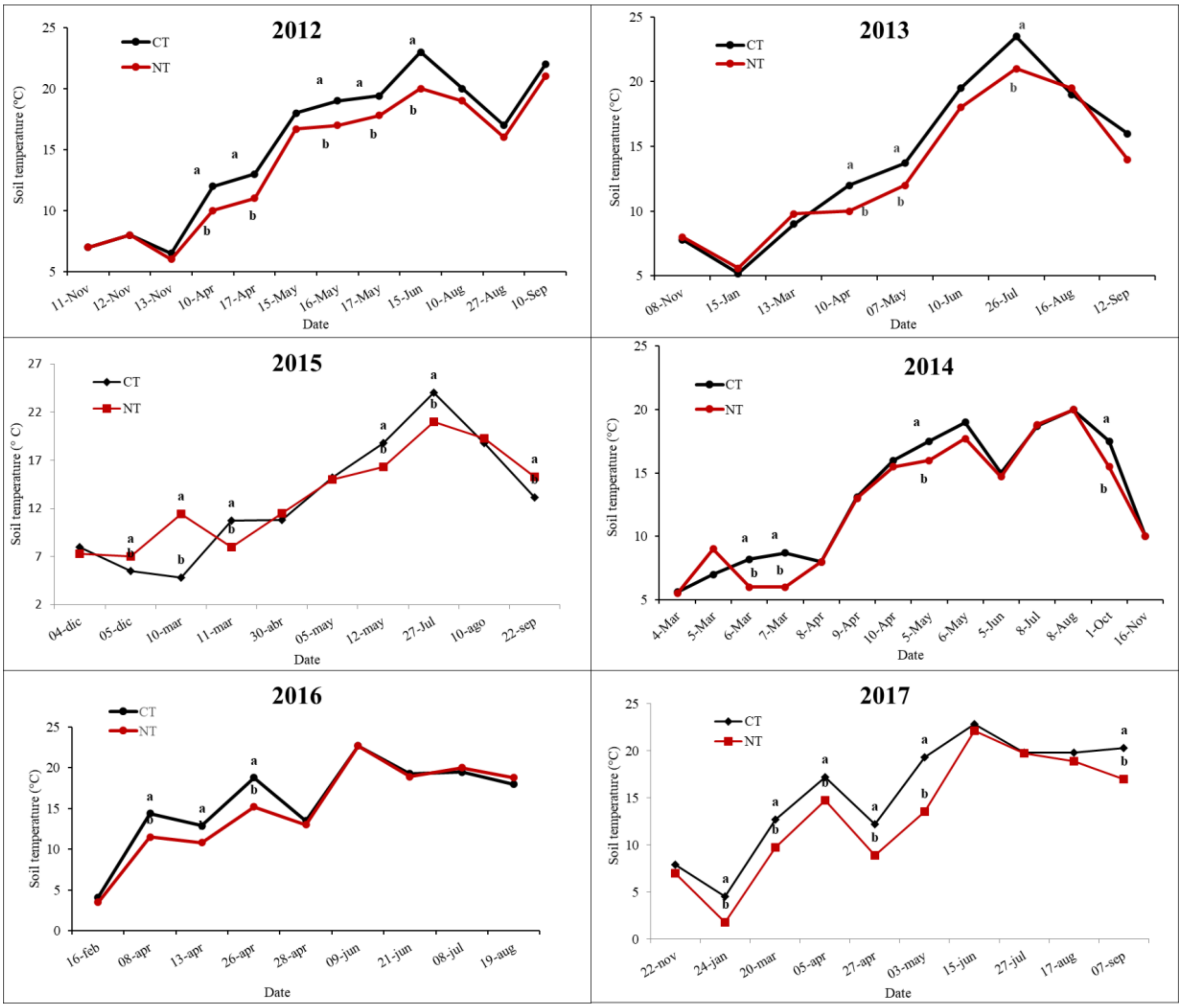

Figure 4.1. Soil temperature from tillage to maize maturity under conventional tillage (CT) and no-tillage (NT) through 2012-2017. Data with different letter are significantly different ($\alpha = 0.05$).

The highest soil moisture level was recorded in May 2015, July 2016 and 2017. After the month of August, this parameter decreased until the soil dried in September (Figure 4.2). The high level of soil moisture during summer months is caused by irrigation treatments that started in May- June and ended in September. The NT system displayed the highest soil moisture levels and went from 1.8 to 7.7%, from 1.3 to 6.3% and from 1.3 to 6.0% greater than CT system in 2015, 2016 and 2017 respectively.

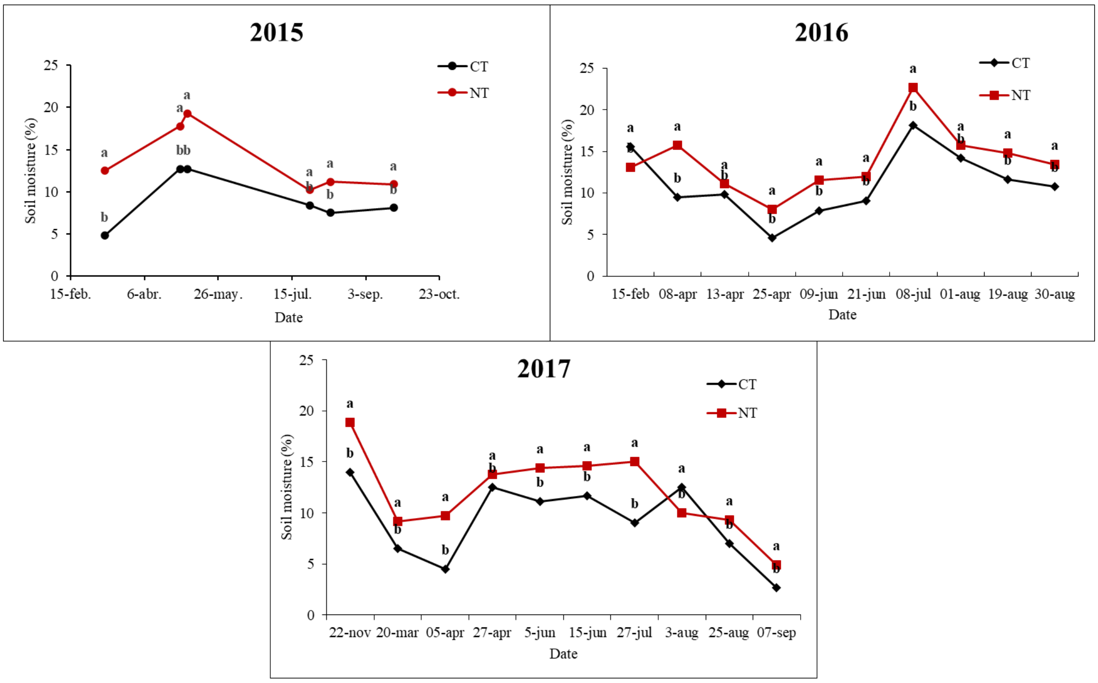

Figure 4.2. Soil moisture from tillage to maize dough stage under conventional (CT) and no-tillage (NT) from 2015 to 2017. Data with different letter are significantly different (α=0.05).

4.3.3. Grain yield and crop residues

The maize grain yield ranged from 9.4 to 17.3 Mg ha^{-1} under CT system and from 11.2 to 19.6 Mg ha^{-1} under NT system. The crop residues left on the soil surface varied from 4.6 to 24.6 Mg ha^{-1} and from 4.6 to 30.2 Mg ha^{-1} under CT and NT systems respectively (Table 4.2). Mean grain yield and crop residues were 6.6 and 17.8 % higher under NT than CT management respectively. Nevertheless, both grain yield and maize residues did not display significant differences among tillage systems during the studied years, except in 2013 and 2017, where the maize grain yield was 16 and 19% significantly higher under NT than CT treatment, respectively.

Table 4.2. Maize grain yield and crop residues under conventional tillage (CT) and no-tillage systems from 2012 to 2017.

	Grain yield (Mg ha^{-1})				Crop residues (Mg ha^{-1})			
Years	Tillage system							
	CT		NT		CT		NT	
2012	17.3	a	18.2	a	24.6	b	30.2	a
2013	16.9	b	19.6	a	11.4	b	15.8	a
2014	15.3	a	14.7	a	9.8	a	8.4	a
2015	16.2	a	16.7	a	4.6	b	8.4	a
2016	14.3	a	14.9	a	10.2	a	10.2	a
2017	9.4	b	11.2	a	5.2	a	4.6	a

Data with the same letter within a row are not significantly different (α = 0.05).

4.3.4. SOC distribution and accumulation

Throughout the 0-30 cm soil depth, the SOC content increased under both tillage systems in 2017 compared to the initial situation in 2011 (Table 4.3). In November 2011, the results showed that mean SOC accumulation was significantly higher in NT than CT plots through the studied soil profile. In the first 30 cm soil depth, SOC content was 24% higher in NT plots than in CT plots. In 2013, the SOC content stored in the soil did not vary significantly according to tillage system; however, it was 0.8, 0.3 and 1.0 Mg ha^{-1} higher under NT than CT system at 0-10, 10-20 and 20-30 cm soil depth respectively. In this year, SOC values were also 8% higher under NT system than the mean obtained under CT in the first 30 cm (Table 4.3). In 2015 and 2017, the SOC stocks were 22% and 36% higher respectively under NT system than CT at 0-10 cm soil depth while at 20-30 cm depth, SOC values were 15 and 5 % higher under CT than NT system in both years respectively.

Table 4.3. Soil organic carbon (SOC) accumulation at 0-30 cm soil depth under conventional tillage (CT) and no-tillage (NT) systems from 2011 to 2017.

Years	2011				2013				2015				2017			
	Tillage system															
SOC (Mg ha^{-1})	CT		NT		CT		NT		CT		NT		CT		NT	
0-10 cm	6.7	b	8.6	a	8.7	a	9.5	a	8.4	b	10.3	a	13.5	b	18.3	a
10-20 cm	8.0	b	9.4	a	9.8	a	10.1	a	10.7	a	11.0	a	13.6	a	14.8	a
20-30 cm	7.5	b	9.5	a	9.6	a	10.6	a	9.8	a	8.5	a	13.5	a	12.8	a
0-30 cm	22.2	b	27.6	a	28.1	a	30.2	a	29.0	a	29.8	a	40.6	b	45.9	a

Data with the same letter within row are not significantly different (α = 0.05).

Mean accumulated SOC showed significant differences among years as in 2017, the soil presented 2.1, 1.7 and 1.5 times higher C accumulation at 0-10, 10-20 and 20-30 cm depths respectively, than the initial year 2011. SOC stocks in 2013 and 2015 were significantly higher than the results obtained in 2011; however, the magnitude of SOC values in 2015 were not significantly different from those found in 2013. Mean SOC stocks in the six year-experiment was 25.7 and 7.6 % higher in NT plots than in CT plots at 0-10 and 10-20 cm soil depths.

3.4.5. Tillage effects on short- and long-term CO_2 emissions

Figure 4.3 shows the evolution of CO_2 emissions (g CO_2 m^{-2} h^{-1}) following tillage operations from 2011 to 2017. Before any soil disturbance, soil CO_2 fluxes showed similar values under CT and NT systems. However, immediately after the soil ploughing in CT system, CO_2 emissions presented an important increase compared to NT system. Under CT, the CO_2 flux measured immediately after tillage ranged from 0.8 to 3.4 g CO_2 m^{-2} h^{-1} in 2017 and 2016, respectively. After the mouldboard ploughing in 2014 and 2016, CO_2 emissions were greater than the results obtained during the other years. In NT plots, soil CO_2 flux was low and stable in this study period. After passing the cultivator, CO_2 emissions presented an increase under CT system and a second peak of the CO_2 flux was observed, except in 2012 and 2014 (Figure 4.3). CO_2 emissions reached 1.1 g CO_2 m^{-2} h^{-1} in 2013 and 0.4 g CO_2 m^{-2} h^{-1} in 2017 under CT system and ranged from 0.1 to 0.3 g CO_2 m^{-2} h^{-1} in 2015 and 2016 under NT. During the hours following the cultivator passing, the CO_2 flux decreased under CT system to reach similar values as the ones recorded under NT system.

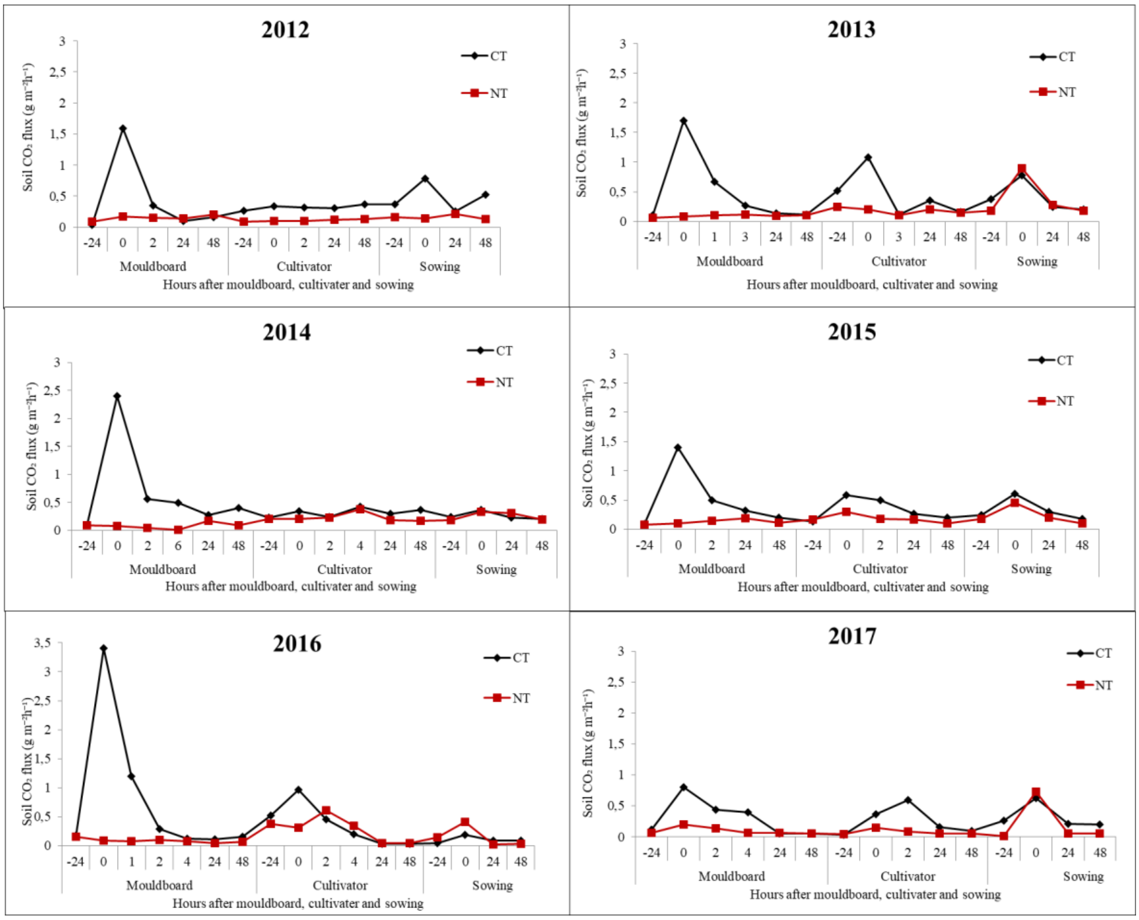

Figure 4.3. CO_2 emissions response to tillage operations (mouldboard, cultivator and sowing) under conventional tillage (CT) and no-tillage (NT) from 2012 to 2017. Data with different letter are significantly different (α=0.05).

Finally, a third peak of the CO_2 flux was observed after sowing under both tillage systems. In 2012 (Figure 4.3), CO_2 emissions presented significant differences between treatments where CT plots had an immediate increase in soil CO_2 flux compared to NT system. The cumulative CO_2 emissions during the first 48 hours after the mouldboard ploughing was significantly higher in CT plots than in the NT plots during the six campaigns studied (Table 4.4). The CO_2 emissions produced from the plots with the mouldboard plough were 3.0 (December 2017) to 10.6 (March 2016) times higher than in NT plots in which the soil was not disturbed.

Table 4.4. Cumulative CO_2 emissions (kg CO_2 ha^{-1}) during the first 48 hours after mouldboard ploughing, cultivator and sowing in conventional tillage (CT) and non-tillage (NT) during the 6-years study.

	Mouldboard				Cultivator				Sowing			
	CT		NT		CT		NT		CT		NT	
2012	1056	a	317	b	643	a	216	b	744	a	230	b
2013	1166	a	182	b	821	a	312	b	590	a	653	a
2014	1726	a	175	b	634	a	408	b	379	a	398	a
2015	1162	a	259	b	734	a	355	b	523	a	360	a
2016	2436	a	182	b	646	a	415	b	178	a	221	a
2017	643	a	216	b	586	a	158	b	494	a	398	a

Data with the same letter within a row are not significantly different ($\alpha = 0.05$).

Cumulative CO_2 fluxes in the first 48 hours after the cultivator pass were significantly higher in CT plots than in NT plots in all the campaigns studied. During the 48 hours after sowing, the cumulative CO_2 flow did not present significant differences among tillage systems except in 2012 when values were significantly higher under CT than NT. In this case, mean CO_2 fluxes in CT and NT plots were 0.48 and 0.38 Mg CO_2 ha^{-1} respectively. Considering the period of 48 h and all the study years, the CO_2 fluxes means were 1034, 350 and 108 kg CO_2 ha^{-1} higher with mouldboard, cultivator and sowing respectively under CT than NT system (Table 4.4).

Figure 4.4 summarizes CO_2 fluxes during the crop cycle and shows that mean CO_2 emissions were higher during the maize reproductive growth stages (R1-R5) under both tillage treatments. Under CT system, the different stages R2 (2012), R3 (2013), V8 (2014), R3 (2015), R1 (2016) and R1 (2017) displayed maximum rates of 0.52, 0.63, 0.62, 0.43, 0.76 and 0.76 g CO_2 m^{-2} h^{-1} while under NT they reached 0.37, 0.58, 0.48, 0.59, 0.64 and 0.50 g CO_2 m^{-2} h^{-1}. During the growing season, CT system had higher CO_2 emissions than NT in all the studied years, but the differences between these treatments were smaller and not always statistically significant, except at R2 in 2012, V8 in 2014, V5 and V8 in 2016 and R1 and R5 in 2017 where CT had significantly higher CO_2 fluxes than NT (Figure 4.4).

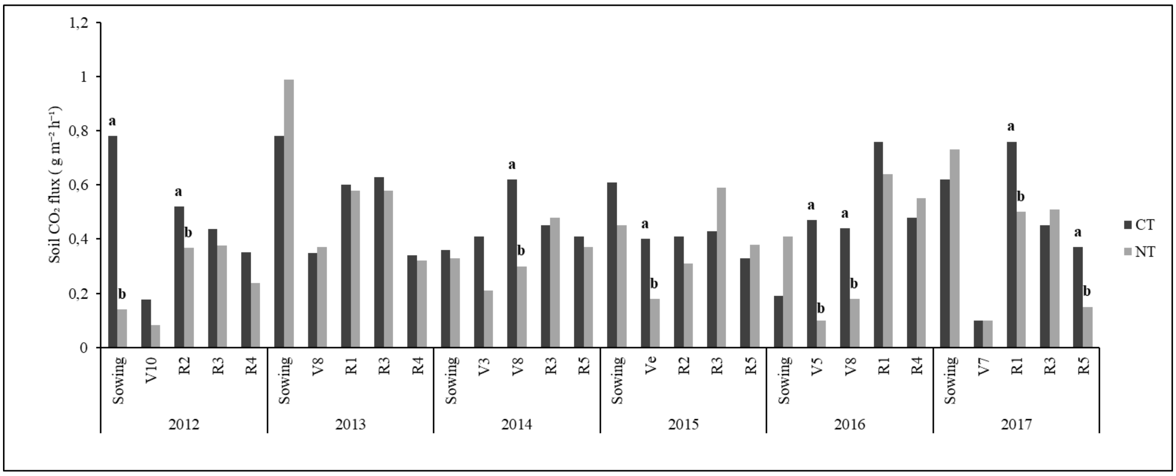

Figure 4.4. CO_2 flux (Mg ha^{-1}) during the maize growing cycle under conventional tillage (CT) and no-tillage (NT) system from 2012 to 2017. Ve: emergence, V3, V5, V7, V8, V10: 3rd, 5th,7th, 8th, 10th leaf developed, respectively. R1: stigma emergence, R2: rennet, R3: milky grain, R4: pasty grain, R5: dented grain. Data with different letter within the same stage are significantly different (α=0.05).

Linear regression analysis of CO_2 fluxes and soil temperature under both tillage system (Table 4.5) showed that CO_2 emissions were significantly affected by temperature in 2013 (R^2 = 0.84**) and 2014 (R^2 = 0.56*) under NT and in 2013 (R^2 = 0.63*) and 2016 (R^2 = 0.58**) under CT. The lack of relationship between soil temperature and gas emissions in the other years could be due to minor temperature variations in some measurements and the different dates when these values were recorded. Soil CO_2 emissions were not affected by soil moisture, except in 2015 where the effects of soil moisture on CO_2 fluxes were significant (CO_2 flux = 0.59 – 0.02 Moisture $_{soil}$, P = 0.03) and accounted for 71% of variability for NT, probably due to major moisture variations between both tillage systems in all measurements. In addition, soil moisture was significantly higher under NT than CT along the maize cycle in 2015 (Figure 4.2).

Table 4.5: Linear regression of CO_2 fluxes and soil temperature and moisture under conventional tillage (CT) and no-tillage (NT).

Year	Tillage system	Mean soil temperature (ºC)	Regression model	R^2	*p* value
2012	CT	18.5	$CO_{2\ flux}$ = 0.72 - 0.017 T_{soil}	0.04	ns
	NT	16.5	$CO_{2\ flux}$ = 0.06 + 0.008 T_{soil}	0.14	ns
2013	CT	14.7	$CO_{2\ flux}$ = 0.03 - 0.044 T_{soil}	0.63*	0.01
	NT	13.7	$CO_{2\ flux}$ = -0.01 + 0.03 T_{soil}	0.84**	0.003
2014	CT	13.2	$CO_{2\ flux}$ = 1.68 + 0.13 T_{soil}	0.24	ns
	NT	12.6	$CO_{2\ flux}$ = 0.17 + 0.16 T_{soil}	0.56*	0.02
2015	CT	14.5	$CO_{2\ flux}$ = 0.17 + 0.02 T_{soil}	0.44	0.07
	NT	13.8	$CO_{2\ flux}$= 0.11 + 0.014 T_{soil}	0.34	0.07
2016	CT	13.6	$CO_{2\ flux}$ = -0.05 + 0.03 T_{soil}	0.58**	0.006
	NT	12.4	$CO_{2\ flux}$ = 0.35 + 0.07 T_{soil}	0.16	ns
2017	CT	14.9	$CO_{2\ flux}$ = 0.12 + 0.02 T_{soil}	0.25	ns
	NT	12.4	$CO_{2\ flux}$ = -0.01 + 0.02 T_{soil}	0.43	0.05
Year	**Tillage system**	**Mean soil moisture (mm)**	**Regression model**	R^2	*p* > F
2015	CT	9.0	$CO_{2\ emissions}$= 0.21 - 0.01 M_{soil}	0.11	ns
	NT	13.7	$CO_{2\ emissions}$= 0.59 - 0.02 M_{soil}	0.71*	0.03
2016	CT	11.2	$CO_{2\ emissions}$= 0.35 + 0.01 M_{soil}	0.02	ns
	NT	13.8	$CO_{2\ emissions}$= 0.08 + 0.03 M_{soil}	0.34	ns
2017	CT	9.5	$CO_{2\ emissions}$= 0.23 + 0.01 M_{soil}	0.09	ns
	NT	11.98	$CO_{2\ emissions}$= 0.31 - 0.01 M_{soil}	0.03	ns

T_{soil}, soil temperature; M_{soil}, soil moisture

The cumulative CO_2 flux (Mg ha^{-1}) measured from sowing to maize maturity and the CO_2 flux / grain yield ratio under CT and NT from 2012 to 2017 are presented in Table 4.6. It can be noticed that under CT and NT, mean cumulative CO_2 fluxes were 14.3 and 11.0 Mg CO_2 ha^{-1} respectively. The ratio of CO_2 emission to grain yield ranged from 0.64 (2013) to 1.41 (2014) under CT and from 0.49 (2012) to 1.15 (2014) under NT (Table 4.6). In this study period, there were significant differences between tillage systems, mean ratio of CO_2 emission to grain yield was 39% significantly lower under NT than under CT, indicating that the amount of CO_2 emission per unit grain decreased under NT.

Table 4.6. Cumulative CO_2 flux (Mg ha^{-1}) from sowing to maturity of maize crop and CO_2 flux / grain yield ratio under CT and NT from 2012 to 2017.

	Soil CO_2 flux (Mg ha^{-1})		CO_2 flux / grain yield	
Year	Tillage system			
	CT	NT	CT	NT
2012	13.9 a	8.9 b	0.8 a	0.5 b
2013	10.7 a	10.4 a	0.6 a	0.5 b
2014	21.6 a	16.9 b	1.4 a	1.2 b
2015	13.5 a	11.5 b	0.8 a	0.7 b
2016	14.5 a	10.4 b	1.0 a	0.7 b
2017	11.7 a	7.8 b	1.2 a	0.7 b

Data with the same letter within a row are not significantly different ($\alpha = 0.05$).

4.4. Discussion

4.4.1. Soil temperature and moisture

No-tillage system recorded low temperature during all different studied seasons, suggesting that crop residues in NT plots diminished the effect of high air temperatures and the solar radiation (Figure 4.1). However, from December to March 2015, the increase of soil temperature under NT could be explained by the fact that the mean air temperature during these months was generally lower than the reported ones during the same months of the other years (Table 4.1). In this context, the crop residues left on the soil surface could have been involved in buffering the impact of the low air temperature on the soil surface. This result coincides with the one found by Ussiri and Lal (2009) who reported higher soil temperature under NT system from November to March. Soil moisture content was higher under NT than CT (Figure.4. 2), the presence of crop residues on the soil surface in NT plots minimised water losses due to evaporation and surface runoff and increased soil moisture, and this was similar to the results found by Ussiri and Lal (2009) in a cultivated maize crop. The low soil temperature and high moisture obtained under NT (Figure 4.1 and 4.2) are in accordance with the results reported by Moitinho et al. (2013) who stated that the absence of residues and the greater surface exposure of the tilled plots enhanced water evaporation and decreased soil moisture in CT system.

4.4.2. Grain yield and crop residues

During this study, maize grain yield did not display significant differences among years, except in 2013 and 2017 when it was higher under NT system than CT (Table 4.2). During both years, the amount of water provided to the crop was the lowest, especially in 2017. In these conditions and throughout the crop cycle, soil moisture was higher under NT than CT system (Figure 4.2) leading to a lower water stress during the cob formation and grain-filling stages and to an increase of the grain yield under NT management. According to some authors, NT system is the most suitable practice in semiarid conditions as it promotes the water accumulation in the soil and increases its use efficiency for the crop compared to CT, and consequently ameliorates crop productivity (Lampurlanés et al, 2001; Cantero et al, 2003). Brouder et al. (2014) and Vanhie et al. (2015) showed that water stress, during phenological stages that required high water uptakes, affected the grain filling, thus the crop productivity. Thierfelder et al. (2015) reported higher maize grain yield under NT system than CT treatment. On the contrary, in a study carried out in the center of Spain, Salem et al. (2015) observed that maize yield under CT exceeded by 15% the NT yield, although the latter system temporarily retained more water in the soil profile than under CT. During an 8-years study conducted in China, Zhang et al. (2012) studied the effects of tillage system on the productivity of maize crop and their results showed that NT system displayed a 2% less yield than CT. In addition, Al-Kaisi et al. (2004) observed, during a 20-years study in Iwoa, that NT system recorded lower yield than CT system, however, these authors specified that the final crop production was higher under NT system thanks to a lower productivity cost and time savings. Obalum et al. (2011) concluded that tillage system did not significantly affect the sorghum production. In 2012, 2013 and 2015, the crop residues were higher under NT than CT system (Table 4.2). This is in accordance with the results reported by Das et al. (2015) and Pittelkow et al. (2015), who agreed that NT produced a greater amount of plant biomass during the development of the crop thanks to a greater water availability and to which crop residues impede the loss of water keeping the soil cooler during the summer. The results obtained in this study demonstrated that maize yield achieved under NT system is comparable and could be higher than the one under CT system in this semiarid region of Castile and Leon.

4.4.3. SOC distribution and accumulation

The results obtained in Table 4.3 showed an increase of the SOC throughout the studied years under both tillage systems. This increase could be explained by the fact that OM acts as a reservoir of nutrients for the crop, participating in the soil biological activity, which lead to a quantitative and qualitative changes of the structure due to tillage (Roldan et al., 2005). Generally, the biological and biochemical parameters of the soil play an important role as early sensitive indicators to ecological stress and soil restoration (Roldan et al., 2003, Izquierdo et al., 2003). During all the studied years, NT system presented significantly higher values than CT at 0-10 cm soil depth, except in 2013 where tillage system did not affect significantly the SOC values at the same depth. Actually, in November 2013, when the soil samples were collected (Table 3), low temperature and precipitation could have caused a low activity of the soil microorganisms, which decreases the OM degradation, and this could explain the absence of the significant difference between tillage systems. The differences found among tillage system was also reported by Huang et al. (2015) who studied the long-term effects of tillage system on different parameters of soil quality in a maize monoculture and found that concentration of OM increased by 18% in the first few centimeters of soil with NT and the amount was associated with the soil aggregates. The crop residues left on the soil surface acted as a protective layer that prevented from losses through evapotranspiration, water and wind erosion and increased the OM content that contributed to improve the soil quality (Traore et al., 2007; Blanco-Canqui, 2013; Zhang et al., 2018). Basamba et al. (2006) and Zhang et al. (2012) pointed out the importance of the accumulation of OM in the upper soil horizon as it improved the quality of the interface between soil and atmosphere and gave the soil greater resistance to different degradation processes that occur on the surface. The results reported by Varvel and Wilhem (2008), Wen-Guang et al. (2015) and Nie et al. (2016) supported the ones obtained in this study, as they found that NT maize led to an accumulation of SOC at or near the soil surface while different tillage treatments including chisel, disk or plough displayed lower SOC values. The crop residues left on the soil surface under NT had a considerable influence on SOC increase at 10 cm depth in 2015 and 2017 (Table 4.4). In CT plots, the mouldboard plough broke the soil structure and SOC content could have been lost due to mineralization. This process did not occur under NT system where the absence of soil disturbance promoted the soil stabilization and the greater accumulation of SOC on the surface. Tillage affects SOC stocks in the ploughed layer by distributing crop residues mechanically throughout the tillage zone (Yang and Wander, 1999) which caused low rates for mineralization distribution of crop residues and homogenization of SOC stocks in the ploughed layers.

In this area, maize crop grows with high temperature and soil moisture (from irrigation) which could lead to high activity of the soil microorganisms promoting a rapid degradation of SOC. However, because of the lack of water during the whole crop cycle in 2017, soil microorganisms' activity decreased and the SOC content was significantly higher than other years. These results coincided with Dimassi et al. (2014) who concluded that the climate interacted with tillage, leading to a greater C sequestration in dry than in wet regions. These authors found that SOC changes under the reduced tillage over time were negatively correlated with the water balance, indicating that sequestration rate was positive in dry periods and negative in wet conditions. De Bona et al. (2006) reported that irrigation increased the decomposition rate of OM by 19% and 15% under CT and NT systems respectively after 8 years of research. Luo et al. (2010) found an increase of the soil C in the topsoil (0-10 cm) under NT but no significant difference was reported over the soil profile to 40 cm, because of the C redistribution through the soil profile under CT system.

4.4.4. Tillage effects on short- and long-term CO_2 emissions

In CT plots, the CO_2 emissions were significantly higher than in NT plots due to soil inversion by mouldboard ploughing that activated the rapid oxidation processes, decreasing the levels of OM in the soil, releasing a large amount of CO_2 into the atmosphere, and contributing to a greater global warming than in NT system. The results obtained in this study (Figure 4.3) coincide with those obtained by Reicosky et al. (1997) who recorded a CO_2 flux that ranged from 0.7 to 2.2 g CO_2 m^{-2} h^{-1} under NT and CT systems respectively in a sorghum monoculture after the mouldboard ploughing. Al-Kaisi and Yin (2005) reported lower soil CO_2 emissions in NT compared with mouldboard plough with the greatest differences occurring immediately after tillage operations in maize-soybean rotation. CO_2 emissions displayed an important increase in the tilled soil after the mouldboard plough use (Figure 4.3) compared to the non-tilled. These results are in accordance with the ones obtained by Prior et al. (2000) who indicated that CO_2 flux increases after the soil ploughing and that it depends on both tillage depth and the degree of soil alteration. The mouldboard ploughing caused aggregates disruption leading to the exposure of the C, previously protected within these aggregates, to the microbial action (Six et al., 2000), resulting in higher CO_2 emissions values under CT when compared to NT treatment. In addition, aggregates disintegration improves soil aeration, so that higher soil CO_2 emissions under CT were related to the higher number of macropores under this management (Silva et al., 2019). The CO_2 flux decreased more than four times considerably in the first 2 hours after the mouldboard pass under CT in all the studied years (Figure 4.4). Reicosky et al. (1997) observed a decrease in the first two hours after the mouldboard pass. After 2 hours, the flux began to decrease until reaching similar values in both treatments at 24 h. Other studies observed that the measurements of CO_2 emissions during short periods after tillage were significantly lower under NT than CT (Alvarez et al., 2001; Alvaro-Fuentes et al., 2007; Carbonell-Bojollo et al. 2011). The results obtained showed that soil tillage operations accelerate CO_2 emissions and the soil C losses (Rakotovao et al., 2017).

The CO_2 emissions reached their maximum in July and August, from the vegetative phenological stages of the crop (V3-V10) to flowering- filling stages (Figure 4.4). At these growth stages, CO_2 emissions were greater under CT than in NT system. The absence of crop residues under CT system induced the increase of soil temperatures and promoted the rapid oxidation of OM. In addition, the microbiological and radicular activity in the soil increased, generating oxidation reactions that resulted in higher CO_2 emissions. Aon et al., (2001) reported that high temperatures resulted in higher decomposition rates of OM. Kuzyakov and Domanski (2000) found that the crop growth had a significant impact on microbial activity through root exudates, and soil microorganisms easily broke them down. This could explain the increase in soil CO_2 emissions observed from the leaf development to the silkimg stage (R1). Hanson et al. (2000) pointed out that for annual crop the contribution of the root to soil respiration are higher during the crop growth and lower during the periods of inactivity. In maize crop, the contribution of rhizosphere respiration (root respiration plus decomposition of root exudates) to total soil respiration can be significant with values close to 50% around the period of maximum crop activity (Rochette et al., 1999). The CO_2 flux ranged from 10.7 (2013) to 21.6 (2014) Mg ha^{-1} under CT tillage and from 7.8 (2017) to 16.95 (2014) Mg ha^{-1} in NT plots from sowing to phenological maturity (Table 4.6). The lower CO_2 emissions under NT system compared to CT could be attributed to a greater surface of crop residues, which could serve as barrier for CO_2 emissions from soil to the atmosphere and reducing soil temperature (Omonode et al., 2007). The slower decomposition of crop residues placed on the soil surface under NT could lead to lower CO_2 fluxes in NT soil

(Curtin et al., 2000). The lower CO_2 emissions in NT plots results agree with those obtained by Reicosky and Archer (2007) and Almaraz et al. (2009). Because of the earlier maturity of maize crop and the lower amount of irrigation water in 2017, CO_2 emissions decreased compared to the other years. The differences of CO_2 emissions reported among the studied years could be caused by the SOC different concentrations in the upper soil layer between years, changes in soil physical processes or soil temperatures and moisture variability.

Rochette et al. (1999) indicated that CO_2 emissions were related to temperature and crop growth and that the highest CO_2 emissions in the warmer months could be associated with root respiration, as the plant growth was also much higher during these months. A significant relationship between CO_2 emissions and both soil temperature and moisture (R^2 = 0.60) were detected under both tillage systems in 2015 and could be caused by the high amount of water applied during irrigation. Omonode et al (2007) found a weak significant relationship between CO_2 emission and soil temperature and moisture when applying different tillage treatments. Under NT, the surface residues provide a barrier between the soil and the atmosphere, which reduces soil evaporation leading to the increase of soil moisture and affects the microbial mobility and gas diffusion in the soil. The general low relationship between CO_2 emissions with soil temperature and moisture found in this study was consistent with other reports (Alkaisi and Yin, 2005; Omonode et al, 2007).

In the six studied years, there were significant differences between tillage systems, CO_2 flux was higher under CT than under NT, except in 2013 (Table 4.6). In this year, the non-significant difference between treatments could be explained by three hypotheses: (1) the combination of the higher amount of crop residues left on the soil surface of 2012 compared to other years, and of the significant increase of grain yields in NT obtained in 2013 (Table 4.2). (2) In 2013, despite the lower water amount irrigated, soil moisture was higher under NT (Rodríguez-Bragado., 2015), which displayed higher yield than CT and led to similar CO_2 fluxes in both treatments. (3) There was a strong correlation between CO_2 emission and soil temperature in 2013 (R^2 = 0.78**) across both tillage systems, while in other years this correlation was lower.

The ratio of CO_2 emissions to grain yield was low under NT system, which means that the amount of CO_2 emissions per unit grain decreased under this practice**.** Under this soil management, CO_2 emissions decreased due to the amount of crop residues left on the soil surface which led to low soil temperatures and high soil moisture (Figure 4.1 and 4.2). In these conditions, the results obtained could be explained by the changes occurring in the root level and microorganism activities, thus enhancing the increase of SOC sequestration at 0-30 cm laying and the decrease of CO_2 emissions.

4.5. Conclusion

This study was initiated to assess SOC stocks, to observe the grain yield response and to determine short-term and seasonal soil CO_2 fluxes under CT and NT practices in continuous maize cropping system during six years in a semiarid region of Castile and Leon. The results showed that SOC stock was 36% greater under NT (18.3 Mg C ha^{-1}) than under CT (13.5 Mg C ha^{-1}) with a rate of 1.61 and 1.13 Mg ha^{-1} yr^{-1} respectively at 0-10 cm depth. In the lower layers, SOC values were 7 and 3 % higher in NT plots than in CT plots at 10-20 and 20-30 cm depths. These results suggest that tillage accelerated the decomposition in the 0-10 cm depth but had minimal influence in the 10-30 cm depth. In 2013 and 2017, maize grain yield reached higher values under NT system than CT, this demonstrates that the non-disturbance of the soil promotes moisture retention for a longer period compared to soil disruption. Generally, this study confirmed that NT management could lead to equal and even higher grain yield than CT system. Short-term CO_2 emissions measured for 48 h after ploughing and cultivator labor were higher under CT than for NT for all measurements. On a seasonal basis, mean CO_2 emissions during the growing season were affected by tillage systems and were 3.32 Mg CO_2 ha^{-1} greater under CT than NT system. Soil temperature and moisture effects on CO_2 flux did not show any significant difference except in 2013 and 2014 under NT and 2015 under CT. Generally specific correlations of soil temperature and moisture with CO_2 emission was insignificant and depended on the moment when the measurements were carried out. From the obtained results in this study, it can be pointed out that the conversion from CT to NT system in irrigated maize would increase the sequestration of OC in the soil and reduce CO_2 emissions to the atmosphere, which can contribute positively to the reduction of GHGs emissions by the agricultural sector without compromising the grain yield.

References

Almaraz, J. J., Zhoub, X., Mabood, F., Madramootoo, C., Rochette, P., Ma, B.L., Smith, D.L. 2009. Greenhouse gas fluxes associated with soybean production under two tillage systems in southwestern Quebec. Soil & Tillage Research, 104(1): 134-139.

Al-Kaisi, M., Yin, X. 2005. Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotations. Journal of Environmental Quality, 34: 437-445.

Al-Kaisi, M., Licht, M.A. 2004. Effect of strip tillage on corn nitrogen uptake and residual soil nitrate accumulation compared with no-tillage and chisel plow. Agronomy Journal, 96(4): 1164 -1171.

Álvaro-Fuentes, J., Cantero-Martínez, C., López, M.V., Arrúe, J.L. 2007. Soil carbon dioxide fluxes following tillage in semiarid Mediterranean agroecosystems. Soil & Tillage Research, 96: 331-341.

Alvarez, R., Alvarez, C.R., Lorenzo, G. 2001. Carbon dioxide fluxes following tillage from a mollisol in the Argentine rolling Pampa. European Journal of Soil Biology, 37: 161–166.

Aon, M.A., Sarena, D.E., Burgos, J.L., Cortassa, S. 2001. Interaction between gas exchange rates and physical and microbiological properties in soils recently subjected to agriculture. Soil and Till Research, 60: 163–171.

Balota, E.L., Kanashiro, M., Filho, A.C., Andrade, D.S., Dick, R.P. 2004. Soil enzyme activities under long-term tillage and crop rotation systems in subtropical agroecosystems. Brazilian Journal of Microbiology, 35: 300-306.

Basamba, T.A., Amezquita, E., Singh, B.R., Rao, I.M. 2006. Effects of tillage systems on soil physical properties, root distribution and maize yield on a Colombian acid-savanna oxisol. Acta Agriculture Scandinavica. Section B-Soil and Plant Science 56 (4): 255-262.

Blanco-Canquí, H. 2013. Crop residue removal for bioenergy reduces soil carbon pools: how can we offset carbon losses? Bioenergy Research, 6: 358-371.

Brouder, S.M., Gomez-Macpherson, H. 2014. The impact of conservation agriculture on smallholder agricultural yields: A scoping review of the evidence. Agriculture, Ecosystems & Environment, 187: 11-32.

Cantero-Martínez, C., Angás, P., Lampurlanés, J. 2003. Growth, yield and water productivity of barley (*Hordeum vulgare L.*) affected by tillage and N fertilization in Mediterranean semiarid, rainfed conditions of Spain. Field Crops Research, 84: 341-357.

Carbonell-Bojollo, R., González-Sánchez, E.J., Veróz- González, O., Ordóñez-Fernándeza, R. 2011. Soil management systems and short-term CO_2 emissions in a clayey soil in southern Spain. Science of the Total Environment, 409(15): 2929-35.

Curtin, D., Wang, H., Selles, F., McConkey, B.G., Campbell, C.A. 2000. Tillage effects on carbon fluxes in continuous wheat and fallow-wheat rotations. Soil Science Society of American Journal, 64: 2080-2086.

Das, A., Ghosh, P.K., Verma, M.R., Munda, G.C., Ngachan, S.V., Mandal, D. 2015. Tillage and residue mulching effect on productivity of maize (*Zea mays*)-toria (*Brassica campestris*) cropping system in fragile ecosystem. Experimental Agriculture, 51(1): 107-125.

De Bona, F.D., Bayer, C., Bergamaschi, H., Dieckow, J. 2006. Soil organic carbon in sprinkler irrigation systems under no-till and conventional tillage. Revista Brasileira De Ciencia do Solo 30(5): 911-919.

Dimassi, B., Mary, B., Fontaine, S., Perveen, N., Revaillot, S., Cohan, J. 2014. Effect of nutrients availability and long-term tillage on priming effect and soil C mineralization. Soil Biology and Biochemistry, 78: 332-339.

Franzluebbers, A.J., Hons, F.M., Zuberer, D.A. 1995. Tillage induced seasonal changes in soil physical properties affecting soil CO_2 evolution under intensive cropping. Soil & Tillage Research, 34: 41-60.

Halvorson, A.D., Del Grosso, S.J., Reule, C.A. 2008. Nitrogen, tillage, and crop rotation effects on nitrous oxide emissions from irrigated cropping systems. Journal of Environmental Quality, 37: 1337-1344.

Hanson, P.J., Edwards, N.T., Garten, C.T., Andrews, J.A. 2000. Separating root and soil microbial contributions to soil respiration: A review of methods and observations. Biogeochemistry, 48: 115-146.

Huang, M., Liang, T., Wang, L., Zhou, C. 2015. Effects of no-tillage systems on soil physical properties and carbon sequestration under long-term wheat-maize double cropping system. Catena, 128: 195-202.

Izquierdo, I., Caravaca, F., Alguacil, M.M., Roldan, A. 2003. Changes in physical and biological soil quality indicators in a tropical crop system (Havana, Cuba) in response to different agroecological management practices. Environmental Management, 32(5): 639-645.

Kuzyakov, Y., Domanski, G. 2000. Carbon input by plants into the soil. Review. Journal of Plant Nutrition and Soil Science, 163(4): 421-431.

Lal, R. 2004. Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. Science Vol. 304, Issue 5677, pp. 1623-1627.

Lal, R. 2007. Farming Carbon. Soil & Tillage Research, 96: 1-5.

Lampurlanés, J., Angás, P., Cantero-Martínez, C. 2001. Root growth, soil water content and yield of barley under different tillage systems on two soils in semiarid conditions. Field Crops Research, 69: 27-40.

Lenka, N.K., Lal, R. 2013. Soil aggregation and greenhouse gas flux after 15 years of wheat straw and fertilizer management in a no-till system. Soil & Tillage Research, 126: 78-89.

Luo, Z., Wang, E., Sun, O.J. 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, Ecosystems & Environment, 139: 224-231.

MAPAMA, 2019. Encuesta sobre superficies y rendimientos de cultivos. Análisis de las técnicas de mantenimiento del suelo y de los métodos de siembra en España. Ministerio de Agricultura, Alimentación y Medio Ambiente en España.

Moitinho, M.R., Padovan, M.P., Panosso, A.R., La Scala, N. 2013. Effect of soil tillage and sugarcane trash on CO_2emission. Revista Brasileira de Ciência do Solo, 37: 1720-1728.

Nie, X.J., Zhang, J.H., Cheng, J.X., Gao, H., Guan, ZM. 2016. Effect of soil redistribution on various organic carbons in a water- and tillage-eroded soil. Soil & Tillage Research, 155: 1-8.

Obalum, S.E., Amalu, U.C., Obi, M.E., Wakatsuki, T. 2011. Soil water balance and grain yield of sorghum under no-till versus conventional tillage with surface mulch in the derived savanna zone of southeastern Nigeria. Experimental Agriculture, 47(1): 89-109.

Omonode, R.A., Vyn, T.J., Smith, D.R., Hegymegi, P., Gal, A. 2007. Soil carbon dioxide and methane fluxes from long-term tillage systems in continuous corn and corn–soybean rotations. Soil & Tillage Research, 95: 182-195.

Pareja-Sanchez, E., Cantero-Martiez, C., Alvaro-Fuentes, J., Plaza-Bonilla, D. 2019. Tillage and nitrogen fertilization in irrigated maize: Key practices to reduce soil CO_2 and CH_4 emissions. Soil & Tillage Research, 191: 29-36.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M., Woomer, P.L. 1997a. Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use Management, 13: 230-244.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G. 1997b. Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use Management, 83: 65-73.

Pittelkow, C.M., Linquist, B.A., Lundy, M.E., Liang, X., Van Groenigen, K.J., Lee, J., Van Gestel, N., Six, J., Venterea, R.T., Van Kessel, C. 2015. When does no-till yield more? A global meta-analysis. Field Crops Research, 183: 156-168.

Prior, S.A., Reicosky, D.C., Reeves, D.W., Runion, G.B., Raper, R.L. 2000. Residue and tillage effects on planting implement-induced short-term CO_2 and water loss from a loamy sand soil in Alabama. Soil & Tillage Research, 54:197-199.

Pumpanen, J., Kolari, P., Ilvesniemi, H., Minkkinen, K., Vesala, T., Niinisto, S., Lohila, A., Larmola, T., Morero, M., Pihlatie, M., Janssens, I., Yuste, J.C., Grunzweig, J.M., Reth, S., Subke, J.A., Savage, K., Kutsch, W., Ostreng, G., Ziegler, W., Anthoni, P., Lindroth, A., Hari, P. 2004. Comparison of different chamber techniques for measuring soil CO_2 efflux. Agriculture and Forest Meteorology, 123: 159-176.

Rakotovao, N.H., Razafimbelo, T.M., Rakotosamimanana, S., Randrianasolo, Z., Randriamalala, J.R., Albrecht, A. 2017. Carbon footprint of smallholder farms in central Madagascar: the integration of agroecological practices. Journal of Clean Production, 140 (3): 1165-1175.

Reicosky, D.C., Dugas, W.A., Torbert, H.A. 1997. Tillage-induced soil carbon dioxide loss from different cropping systems. Soil & Tillage Research, 41: 105-118.

Reicosky, D.C., Archer, D.W. 2007. Mouldboard plow tillage depth and short-term carbon dioxide release. Soil and Till Research, 94:109-121.

Rochette, P., Flanagan, L.B., Gregorich, E.G. 1999. Separating soil respiration into plant and soil components using analyses of the natural abundance of carbon-13. Soil Science Society of American Journal, 63: 1207-1213.

Rodríguez-Bragado, L. 2015. Efectos del sistema de laboreo sobre las propiedades y calidad del suelo, dinámica del agua de riego y producción del cultivo de maíz. Doctoral Thesis. Univ. Valladolid. ETSIA. Palencia.

Roldan, A., Caravaca, F., Hernandez, M.T., Garcia, C., Sanchez-Brito, C., Velasquez, M., Tiscareno, M. 2003. No-tillage, crop residue additions, and legume cover cropping effects on soil quality characteristics under maize in Patzcuaro watershed (Mexico). Soil & Tillage Research, 72(1): 65-73.

Roldan, A., Salinas-Garcia, J.R., Alguacil, M.M., Diaz, E., Caravaca, F. 2005. Soil enzyme activities suggest advantages of conservation tillage practices in sorghum cultivation under subtropical conditions. Geoderma, 129(3-4): 178-185.

Salem, H.M, Valero, C., Angel Muñoz, M., Gil Rodriguez, M., Silva, L.L. 2015. Short-term effects of four tillage practices on soil physical properties, soil water potential, and maize yield. Geoderma, 237: 60-70.

Silva, B., Moitinho, M.R., Araujo, G.A., Teixeirac, D., Fernandes, C., La Scala, N.J. 2019. Soil CO_2 emission and short-term soil pore class distribution after tillage operations. Soil & Tillage Research, 186: 224-232.

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

Sombrero, A., De Benito, A. 2010. Carbon accumulation in soil. Ten-year study of conservation tillage and crop rotation in a semi-arid area of Castile-Leon, Spain. Soil & Tillage Research, 107(2), 64-70.

Thierfelder, C., Matemba-Mutasa, R., Rusinamhodzi, L. 2015. Yield response of maize (Z*ea mays* L.) to conservation agriculture cropping system in southern Africa. Soil & Tillage Research, 146: 230-242.

Traore, S., Thiombiano, L., Millogo, J.R., Guinko, S. 2007. Carbon and nitrogen enhancement in cambisols and vertisols by acacia spp. in eastern Burkina Faso: Relation to soil respiration and microbial biomass. Applied Soil Ecology, 35(3): 660-669.

Ussiri, D., Lal, R. 2009. Long-term tillage effects on soil carbon storage and carbon dioxide emissions in continuous corn cropping system from an alfisol in Ohio. Soil & Tillage Research, 104: 39-47.

Vanhie, M., Deen, W., Lauzon, J.D., Hooker, D.C. 2015. Effect of increasing levels of maize (*Zea mays L.*) residue on no-till soybean (glycine max merr.) in northern production regions: A review. Soil & Tillage Research, 150: 201-210.

Varvel, G.E., Wilhelm, W. 2008. Soil carbon levels in irrigated western Corn Belt rotations. Agronomy Journal, 100: 1180-1184.

Vinten AJA, Ball BC, O'Sullivan MF, Henshall JK, 2002. The effects of cultivation method, fertilizer input and previous sward type on organic C and N storage and gaseous losses under spring and winter barley following long-term leys. J Agric Sci 139 (3): 231-243.

Wen-guang, T., Xiao-Ping, X., Hai-Lin, Z., Zhong-Du, C., Jian-Fu, X. 2015. Effects of long-term tillage and rice straw returning on soil nutrient pools and Cd concentration. Yingyong Shengtai Xuebao. Vol. 26, Issue 1: 168-176.

West, T.O., Post, W.M. 2002. Soil Organic Carbon Sequestration Rates by Tillage and Crop Rotation: A Global Data Analysis. Soil Science Society of American Journal, 66(6): 1930-1946.

Yang, X., Wander, M.M. 1999. Influence of tillage on the dynamics of loose-and occluded-particulate and humified organic matter fractions. Soil Biology and Biochemistry, 32(8): 1151-1160.

Zhang, X. 2012. Cropping and tillage systems effects on soil erosion under climate change in Oklahoma. Soil Science Society of American Journal, 76(5): 1789-1797.

Zhang, W., He, X., Zhang, Z., Gong, S., Zhang, Q., Zhang, W., Liu, D., Zou, C., Chen, X. 2018. Carbon footprint assessment for irrigated and rainfed maize (*Zea mays L.*) production of the loess plateau of China. Biosystems Engineering, 167: 75-86

CHAPTER 5. EFFECT OF TILLAGE SYSTEMS AND DIFFERENT RATES OF NITROGEN FERTILISATION ON THE CARBON FOOTPRINT OF IRRIGATED MAIZE IN A SEMIARID AREA OF CASTILE AND LEON, SPAIN

Dachraoui, M., Sombrero, A. 2020. Effect of tillage systems and different rates of nitrogen fertilisation on the carbon footprint of irrigated maize in a semiarid area of Castile and Leon, Spain.

Soil & Tillage Research, 196, 104472.

5.1. Introduction

The greenhouse effect is a natural phenomenon resulting from the presence of gases in the atmosphere which absorb the thermal infrared radiation produced by the Earth's surface, preventing the mean global temperature to reach almost -18ºC and keeps it around 15ºC (Seguin and Soussana, 2008). This phenomenon is being emphasized since the industrial revolution with the increase of the carbon dioxide (CO_2), the nitrous oxide (N_2O) and methane emissions (CH_4) in the atmosphere. A century and a half of industrialization, including clear-felling forests and certain farming methods, has driven up quantities of greenhouse gases (GHGs) in the atmosphere. In the mid-2015, the global population has reached 7.3 billion and is expected to reach 9.7 billion by 2050. This increase will significantly contribute to the magnitude of GHGs emissions into the atmosphere (United Nations, 2019). To meet the needs of the rising global population, one of the most concerned sectors is agriculture, for it is the one providing food and raw material to agro-alimentary sectors and other ones. Agriculture is both a producer and consumer of energy. It uses large quantities of locally available energy in elementary activities such as tillage, sowing, harvest and transportation and on a secondary scale in the processes of manufacturing, packaging and fertilizers and pesticides storage. These quantities of energy are used directly and indirectly in the form of diesel, fertilizers for plant protection, chemicals, irrigation water and machinery (Mughal and Amjad, 2012). The use of this energy from alternate sources emits CO_2 into the atmosphere (Lal, 2004a), which is the main contributor to the enhanced greenhouse effect, for it is the gas with the largest radiative effect and the one which has the longest adjustment time to a new equilibrium if sources or sinks change reaching 200 years (IPCC, 2007).

Meanwhile, the nitrous oxide formation in soils occurs mainly through nitrification and denitrification processes, which are influenced by soil moisture, temperature, oxygen concentration, amount of available organic carbon, nitrogen and soil C: N ratio (Signor and Cerri, 2013). The N_2O presents a global warming potential 298 times more effective than CO_2 (IPCC, 2007). Most of the emissions are biogenous and take place in soils. The emission of the N_2O is also affected by soil management practices induced by human activities such as soil tillage, recycling N from crop residues and the application of N-fertilizers. Not only N_2O direct emissions are involved in the greenhouse effect but also indirect emissions produced from the transfer of N fertilizer to the atmosphere through NH_3 volatilisation and from N leaching and runoff (Reay et al., 2005).

The world's soils are estimated to have a high sink potential for carbon sequestration, not only in terms of their large potential of carbon content, but also because soil organic carbon (SOC) is particularly responsive to modification through agricultural land use (Corsi et al., 2012). But the conversion of natural lands to agricultural ecosystems causes depletion of the SOC pool because of the lower return of biomass, the higher losses of SOC by erosion, mineralization and leaching and the stronger variation in soil temperature and moisture regimes (Lal et al, 2015). This depletion is exacerbated when the output of C exceeds the input and when soil degradation is severe (Lal, 2004b). Thus, agricultural soils contain 25-75% less SOC than their counterparts in undisturbed or natural ecosystems. According to Lal et al. (2015), a field of maize (*Zea mays*) can capture about 400 times as much C as the annual increase by anthropogenic emission of CO_2 in the entire column of air above the field from ground to upper reaches of the atmosphere. Therefore, identification and adoption of site-specific soil and crop management systems can lead to the sequestration of atmospheric CO_2 (Gan et al., 2014). One of this management techniques that could provide a net carbon sink in soils is conservation tillage, which minimizes or eliminates manipulation of the soil for crop production. It includes the practice of mulch tillage, which leaves crop residues on the soil surface. These procedures generally reduce soil erosion, improve water use efficiency and fertility, boost soil biodiversity and increase carbon concentrations in the topsoil (Lal, 2004a).

It is considered necessary to re-establish the CO_2 initial balance between the terrestrial surface and the atmosphere CO_2; in one hand by reducing the GHGs effect and on the other hand by incrementing the soil carbon fixation. In comparison with other sectors that contribute to the greenhouse effect, the changes and management practices brought to soil in the agrarian sector has a double synergistic impact, as mitigation (emissions reduction) and as a sink (immobilised carbon increase). So, identifying the carbon footprint (CFP) is an important component of sustainable agriculture (Smith et al., 2008). The CFP of a product is used to quantify the sum of GHGs emissions and removals as CO_2 equivalents (CO_{2eq}) in a product to mitigate climate change (ISO 14067, 2013). These GHGs emissions including the non-CO_2 emissions caused directly and indirectly by human - induced activity such as tillage, fertilization and harvesting affect the intensity of the CFP of an agricultural product. Thus, adjusting farming practices would supply a potential solution for reducing GHGs emissions and would prevent an economic loss for crop producers. Because previous studies focused on assessing the CFP according to different crop patterns (Qi et al., 2018) or as a response to crop rotation systems (Gan et al., 2012; Yang et al., 2014), it was interesting to study the effect of tillage systems and different fertilization rates on the CFP of a mechanized spring maize. Moreover, there was no studies conducted on the analysis of maize CFP in the semiarid conditions of Castile and Leon. Therefore, this work aims to estimate the total emissions of GHGs produced from the agricultural energy inputs, the N_2O emissions and SOC changes in the soil in relation with grain yield expressed in $tCO_{2eq}\ t^{-1}$. The estimation of these components of the CFP is determined under conventional tillage (CT) and no-tillage (NT) using different N fertilization rates in continuous irrigated maize in semi-arid zone of Castile and Leon, Spain.

5.2. Material and methods

5.2.1. Experimental design and crop management

To avoid repetition the experiment design and crop management methodology are provided in ***Chapter 3***

5.2.2. Energy inputs of maize production

The energy inputs included machinery, diesel fuel, electricity, maize seed, chemical fertilizers, herbicides and water supply for irrigation whereas the energy equivalent of grain yield was considered as the energy output. The different inputs and output were multiplied to energy by various coefficients collected from literature as it is shown in Table 5.1. In this study, the human activity was not considered in the energy balance due to its very low percentage of energy input (<0.02%) for production systems in developed countries (Khaledian et al, 2010). The grain drying and the maize residues were not considered either as produced energy because of their incorporation into the soil in CT system or their remaining on the soil surface in NT production.

Table 5.1. Energy equivalent of the different components used in the maize production operations.

	Dose/Unit	Energy equivalent	References
Fertilizers	kg ha^{-1}	MJ.kg.UF	
N		45	Hérnanz, 2007
P_2O_5	800	15.8	Hérnanz, 2007
K_2O		9.3	Hérnanz, 2007
NAC 27%	700, 600	45	Hérnanz, 2007
ENTEC 26%	700, 600	45	Hérnanz, 2007
Herbicides	l ha^{-1}	MJ kg ai^{-1}	
Glyphosate	2.5	474	Audsley et al, 2009
Camix (S-metolachlor + Mesotrione)	3.5	150+691	Audsley et al, 2009
Closar 5 gr (Chlorpyrifos)	15 kg ha^{-1}	324	Audsley et al, 2009
Lontrel (Clopyralid)	0.15	432	Audsley et al, 2009
Emblem (Bromoxynil 20%)	2.25	302	Audsley et al, 2009
Starane (Fluroxypyr 1-methylheptyl ester)	1	518	Audsley et al, 2009
Karate Zeon (10% p/v Lambda Cyhalothrin)	0.1	529	Audsley et al, 2009
Maize seed	kg	15	Hérnanz, 2007
Diesel	l	47.45	Hérnanz, 2007
Electricity	kWh	11.93	Kitani, 1999
Irrigation	m^3	1.02	Erdal et al, 2007

The methodology followed to calculate the direct energy consumed by the tractor to execute the different operations of the maize production and by the use of electricity for the irrigation process was suggested by Hérnanz (2007). The energy consumed by the irrigation system depended on the hydric needs of the maize crop, the impetus of the pump system and the pipeline, was quantified by Khaledian et al (2010). The different amount of the fuel consumed per hour, the associated energy for machinery and electricity in both tillage systems are detailed in Table 5.2.

Table 5.2. Energy inputs of the different field operations and the irrigation equipment in maize production in both tillage systems during all the growing seasons of the study.

Field operations	Diesel (l ha^{-1})		Energy (Kj kg^{-1} h^{-1})		Time (h ha^{-1})		Tool's weight (kg)		Energy (Mj ha^{-1})	
	CT	NT	CT	NT	CT	NT	CT	NT	CT	NT
Mouldboard ploughing	21.8	-	82.6	-	1.58	-	750	-	97.9	-
Field cultivation	4.4	-	60.0	-	0.44	-	600	-	15.8	-
Herbicide Spray (Glyphosate)	-	1.7	-	94.2	-	0.2	-	400	-	6.4
Base dressing fertilization	1.3	1.3	90.0	90.0	0.2	0.2	250	250	4.5	4.5
Maize planting	18.3	18.3	110.0	110.0	1.84	1.8	950	950	192.3	192.3
Herbicide Spray	1.7	1.7	94.2	94.2	0.17	0.2	400	400	6.4	6.4
Herbicide Spray	1.7	1.7	94.2	94.2	0.17	0.2	400	400	6.4	6.4
Top dressing fertilization	1.3	1.3	90.0	90.0	0.2	0.2	250	250	4.5	4.5
Harvest	17.5	17.5	54.0	54.0	2.2	2.2	3492	3492	418.6	418.6
Plant residues chopping	-	10.1	-	61.0	-	0.7	-	500	-	22.6
Tractor			13.1		4.6	3.5	3595		215.8	163.7
Irrigation (Installation)									888.85	
Irrigation (Bomb)									118.16	

5.2.3. Soil organic carbon (SOC) changes

After the maize harvest and before the preparatory work in November 2011, baseline soil samples were collected at three sites in every elementary plot to obtain a mixed sample per plot at depths of 0-10, 10-20, 20-30 cm. These plots were thereafter re-sampled at 2-year intervals in 2013, 2015 and 2017. A total of 120 soil samples were obtained during the 6 years of the study. These samples were air dried and sieved through a 2 mm mesh. Afterward, they were taken to the laboratory where they were analysed to determine the soil organic carbon (SOC) and nitrogen (N) content by dry combustion with a LECO CNS 1934, these analysis results were used in the estimation of CFP. The SOC was calculated in terms of elemental soil mass considering the concentration of the organic carbon, bulk density (Bd) that was similar under both tillage systems.

The assessment of the CFP was based on the Intergovernmental panel on Climate Change, 2006 (IPCC). In the agricultural sector, the greenhouse gases flow balance is estimated by evaluating the change of carbon stocks, which is used to estimate most of CO_2 flows (yet the biggest part of these changes is produced through the CO_2 exchange between the terrestrial surface and the atmosphere) and by evaluating the gas flow from and to the atmosphere (IPCC, 2006). Soil carbon is an important factor in influencing the CFP of the cropping systems as it changed substantially over time. In our study, the annualized soil C gain or loss was determined as follow:

$$\Delta SOC = \frac{SOC_{2017} - SOC_{2011}}{6} \times \frac{44}{12} \quad (1)$$

where ΔSOC is the annual change in SOC since 2011 (tCO_{2eq} ha $^{-1}$ year $^{-1}$); SOC_{2011} and SOC_{2017} are the amount of SOC in the 0-30 cm soil in 2011 and 2017, respectively; 6 is the duration of the study period and 44/12 is the coefficient converting C into CO_2.

5.2.4. Direct and indirect N_2O emissions

The amount of direct and indirect N_2O emissions is related to the quantity of N applied to the crop and is affected by the environmental conditions (Gregorich et al., 2005). Following the model of Rochette et al. (2008) which measures N_2O fluxes from Canadian farmlands, the estimation of direct N_2O emissions factor (EF), in this study, was based on the ratio of growing season precipitation and irrigation (PI) to evapotranspiration (PE):

$$EF = \frac{0.022PI}{PE} - 0.0048 \quad (2)$$

Crop residues were considered as a source of N denitrification and nitrification assuming that the N contained was released as N_2O in the same year of production (Ma et al., 2012). To quantify the N from crop residues, the above and belowground biomass was considered and multiplied by its respective N concentration (IPCC, 2006). Considering all the different factors that affect N_2O direct emissions, the following equation was used:

$$N_2O_{Direct} = (F_{SN} + F_{CR} + F_{SOM}) \times EF \times \frac{44}{28} \times 298 \quad (3)$$

where F_{SN} is the annual amount of synthetic fertilizer N applied to soils (kg N yr^{-1}); F_{CR} is the annual amount of N in crop residues above and below ground (kg N yr^{-1}); F_{SOM} is the annual amount of N in mineral soils that is mineralised, in association with loss of soil C from soil organic matter as a result of changes of land use or management (kg N yr^{-1}); EF is the emission factor for N_2O emissions from N inputs (kg N_2O–N (kg N input)$^{-1}$); 44/28 is the conversion coefficient from N_2O-N to N_2O and 298 is the global warming potential of N_2O for the 100-year period (IPCC, 2006).

Soil mineral N, particularly under the form of nitrate in the rooting zone, is subject to leaching (Campbell et al., 2004), and this N can be leached out of the rooting zone and/or undergone further transformations to be emitted as N_2O (Gan et al., 2012). The fraction of N subject to leaching ($FRAC_{LEACH}$) was estimated as the following equation under CT management, while it was 21% lower under NT treatment (Goss and Howse, 1993):

$$FRAC_{LEACH} = \frac{0.324\ PI}{PE} - 0.0247 \quad (4)$$

Using the method developed by the Intergovernmental Panel on Climate Change (IPCC, 2006) indirect emissions of N_2O produced from N leaching were estimated as:

$$N_2O_{LEACH} = (F_{SN} + F_{CR} + F_{SOM}) \times FRAC_{LEACH} \times EF_{LEACH} \times \frac{44}{28} \times 298 \quad (5)$$

where F_{SN} is the annual amount of synthetic fertilizer N applied to soils in regions where leaching occurs (kg N yr^{-1}); F_{CR} is the amount of N in crop residues above and below ground (kg N yr^{-1}); F_{SOM} is the annual amount of N mineralised in mineral soils associated with loss of soil C from soil organic matter as a result of changes to land use or management in regions where leaching/runoff occurs (kg N yr^{-1}); $FRAC_{LEACH}$ is the fraction of applied N/mineralised N by loss of leaching; EF_{LEACH} is the emission factor for N_2O emissions from N leaching (0.0075); 44/28 is the conversion coefficient from N_2O-N to N_2O and 298 is the global warming potential of N_2O for the 100-year period.

The amount of indirect N_2O emissions produced from atmospheric deposition of N volatilised as NH_3 and NO_x from the synthetic fertilization was also considered in this study as:

$$N_2O_{VD} = F_{SN} \times FRAC_{GASF} \times EF_{VD} \times \frac{44}{28} \times 298 \quad (6)$$

where F_{SN} is the annual amount of synthetic fertilizer N applied to soils (kg N yr^{-1}); $FRAC_{GASF}$ is the fraction of synthetic fertilizer N that volatilises as NH_3 and NOx volatilised and is considered 0.1; EF_{VD} is the emission factor for N_2O emissions from atmospheric deposition of N on soils surfaces and is considered 0.01 in this study.

The sum of the N_2O emissions from N leaching and from the deposition of N volatilisation constituted the indirect N_2O emissions.

5.2.5. *Assessment of the maize carbon footprint*

Based on the experimental data the CFP of maize under CT and NT systems was determined by the total GHGs emissions and grain yield. The total GHGs emissions included the GHGs emissions resulting from agricultural inputs and the non-CO_2 emissions (N_2O emissions) with and without consideration of the changes in SOC storage during the experiment (Gan et al., 2012; Zhang et al., 2016).

$$CFP = \frac{E_{inputs+} E_{N2O+} \Delta SOC}{Y} \quad (7)$$

where CFP is the carbon footprint of the maize production (tCO_{2eq} t^{-1}); Y is the grain yield (tons); E_{inputs} the total amount of GHGs emissions due to the application of agricultural inputs (tCO_{2eq} ha^{-1}); E_{N_2O} is the cumulative of N_2O direct and indirect emissions from the maize crop (tCO_{2eq} ha^{-1}), estimated by the IPCC, 2006. ΔSOC is the amount of change in SOC (tCO_{2eq} ha^{-1}).

5.2.6. *Statistical analysis*

The results obtained were statistically analyzed using ANOVA or the general linear model (GLM) procedure (SAS Institute, 9.4) applying Tukey's test at the 5% significant level (*P≤0.05)*.

5.3. Results

5.3.1. *Energy inputs of the maize production*

The energy inputs assigned to every component of the maize production in 2012/2014 and 2015/2017 reflected in Table 5.3, allowed to notice that the application of synthetic fertilizers, the use of electricity and irrigation presented the production factors with the highest percentages reaching 79% and 77% of the total energy inputs under CT and systems respectively. Based on the statistical analysis, CT system was characterised by higher fuel consumption, which was significantly higher than the amount used by NT system by 5.5% of the total direct emissions. This difference, obviously, was maintained in the use of agricultural machinery, which was significantly higher under CT management than NT system. The energy input of pesticides use was highly significant under NT system; it was almost 2.5 and 2 times higher than the energy input under CT system in 2012/2014 and 2015/2017 respectively. In 2012/2014, the application of synthetic fertilizers was significantly higher when using the FC than FR by 2% in both tillage systems. This increase affected the total energy inputs, which were also 2% higher with FC than with FR in both tillage systems. In addition, in 2015/2017, the elevated energy input was attributed to the use of FC that was 1 and 2% higher than FE and FER, respectively in both tillage systems. The use of the reduced rate of N-fertilization (FER) resulted in the lowest total energy inputs under both tillage systems and especially under NT treatment.

Table 5.3. Energy inputs under CT (conventional tillage) and NT (no-tillage) systems and N-fertilisation rates in 2012/2014 and 2015/2017 for the different production components.

		Direct Energy (t CO_{2eq} ha^{-1})			Indirect Energy (t CO_{2eq} ha^{-1})						**Total energy inputs** (t CO_{2eq} ha^{-1})
2012/2014											
Soil Management	N-Fertilization	Diesel	Electricity	Total	Machinery	Fertilizers	Pesticides	Maize seed	Water	Total	
CT	FC	0.24 a	0.37	0.61 a	0.15 a	1.06 a	0.03 b	0.03	0.31	1.59 b	2.19 a
	FR					0.97 b				1.50 d	2.10 c
NT	FC	0.19 b		0.56 b	0.14 b	1.06 a	0.07 a			1.62 a	2.18 a
	FR					0.97 b				1.53 c	2.09 d
2015/2017											
CT	FC	0.24 a	0.43	0.67 a	0.15 a	1.06 a	0.03 b	0.03	0.34	1.62 ab	2.28 a
	FE					1.04 b				1.59 b	2.26 b
	FER					0.96 c				1.51 c	2.18 d
NT	FC	0.19 b		0.62 b	0.14 b	1.06 a	0.07 a			1.64 a	2.25 b
	FE					1.04 b				1.61 ab	2.23 c
	FER					0.96 c				1.54 c	2.15 e

Letters within the same column indicate statistical differences among different treatments ($P \leq 0.05$).
FC, conventional fertiliser NAC27 ; FR, reduced fertiliser NAC27 ; FE, conventional fertiliser ENTEC26 ; FER, reduced fertiliser ENTEC26.

5.3.2. SOC evolution over the studied years

According to Figure 5.1, and due to the absence of differences when considering the N-fertilization rates, mean SOC is only shown according to tillage system. The evolution of SOC stock in the first 30 cm depth of the soil witnessed an increase throughout the study years in both tillage systems, being significantly higher under NT management in 2011 with a mean of 27.5 t C ha^{-1} than under CT system (22.3 t C ha^{-1}). In both 2013 and 2015, the amounts of SOC under both tillage systems did not vary significantly. However, in the last year of the experiment (2017) NT plots recorded higher stock of SOC (44.3 t C ha^{-1}) than CT plots (38.3 t C ha^{-1}).

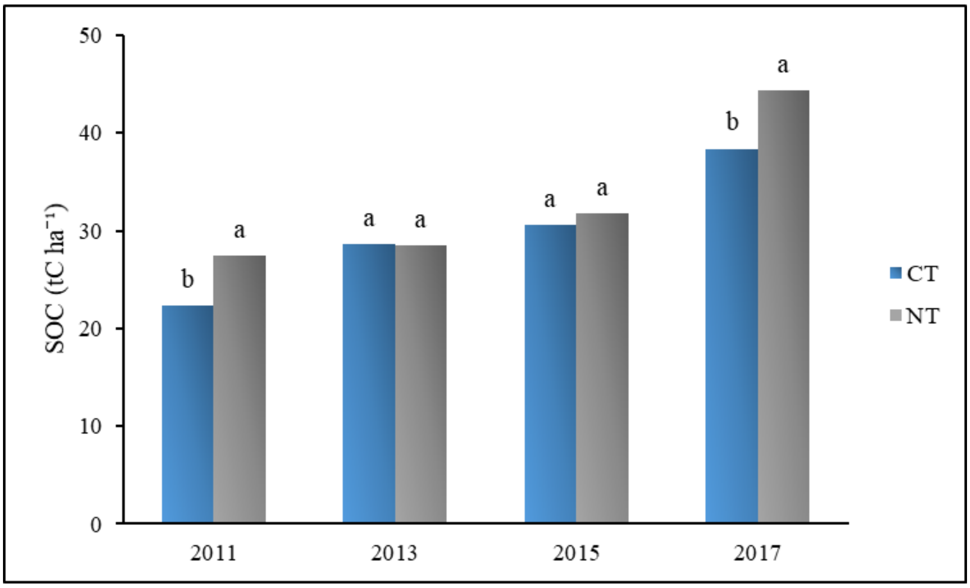

Figure 5.1. Soil organic carbon (SOC) evolution during the 6-year study under conventional tillage (CT) and no-tillage (NT) systems. Letters indicate statistical difference among different treatments ($P \leq 0.05$)

5.3.3. Direct and indirect N_2O emissions

Considering the mean total N_2O emissions (direct and indirect) reflected in Figure 5.2. A, no statistical difference was highlighted under CT and NT systems in 2012/2014. However, the highest emissions were obtained by the application of FC reaching 4.2 and 4.4 8 tCO_{2eq} ha^{-1} under CT and NT respectively while the lowest emissions were produced by the FR application with 3.9 and 3.8 t CO_{2eq} ha^{-1} under CT and NT systems respectively. In 2015/2017 (Figure 5.2. B), N_2O emissions did not show significant differences between tillage systems and the lowest emissions were also recorded under the lowest N-fertilization rate (FER) with 3.3 and 3.4 tCO_{2eq} ha^{-1} under CT and NT managements respectively.

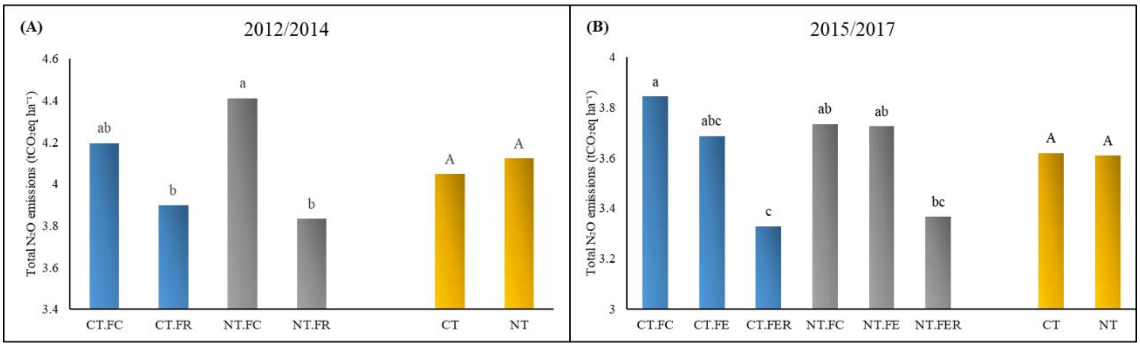

Figure 5.2. Total N_2O emissions produced by CT (conventional tillage) and NT (no-tillage) system in 2012/2014 (A) and 2015/2017 (B). Letters indicate statistical difference among different treatments (*P*≤0.05). FC, conventional fertiliser NAC27; FR, reduced fertiliser NAC27; FE, conventional fertiliser ENTEC 26; FER, reduced fertiliser ENTEC 26.

According to Figure 5.3, direct N_2O emissions generated from nitrification and denitrification processes resulted in the highest N_2O emissions produced during the study. They reached 86% under CT and 88 % under NT system in 2012/2014 (Figure 5.3. A) and they ranged from 85 to 86% under CT and reached 87% under NT system in 2015/2017 (Figure 5.3. B). While indirect N_2O emissions from N leaching displayed lower percentages with 11% under CT plots and 9% under NT plots in both periods of the study (Figure 5.3. C and D). It is necessary to mention that in the current study N_2O emissions produced from atmospheric deposition of N volatilised as NH_3 and NO_x from the synthetic fertilisation was not mentioned because they accounted for 1% of the total N_2O emissions.

Based on statistical analysis, in 2012/2014 direct N_2O emissions did not show significant difference between N-fertilization rates (FC and FR) under CT system (Figure 5.2. A) while the FC-fertilization affected significantly the direct N_2O emissions under NT system which reached 3.9 tCO_{2eq} ha^{-1} in comparison to 3.4 tCO_{2eq} ha^{-1} with FR. Generally, tillage systems did not display statistical differences regarding the N_2O direct emissions. Also, in 2015/2017 (Figure 5.3. B), direct N_2O emissions did not underline differences between tillage systems. However, under CT management, the direct N_2O emissions produced from the use of FC was significantly higher (3.3 tCO_{2eq} ha^{-1}) than the ones produced from FER (2.9 tCO_{2eq} ha^{-1}), while there were no differences observed with the

FE. Direct N_2O emissions produced from the different rates of N-fertilization did not display statistical differences under NT system (Figure 5.2. B). Although, in 2012/2014, direct N_2O emissions were not statistically different, they were slightly higher in NT plots (3.6 t CO_{2eq} ha^{-1}) than in CT plots (3.5 t CO_{2eq} ha^{-1}). The results displayed in Figure 5.3 (C and D) show that CT practice contribute to significantly higher indirect emissions than NT management in 2012/2014 and 2015/2017 respectively.

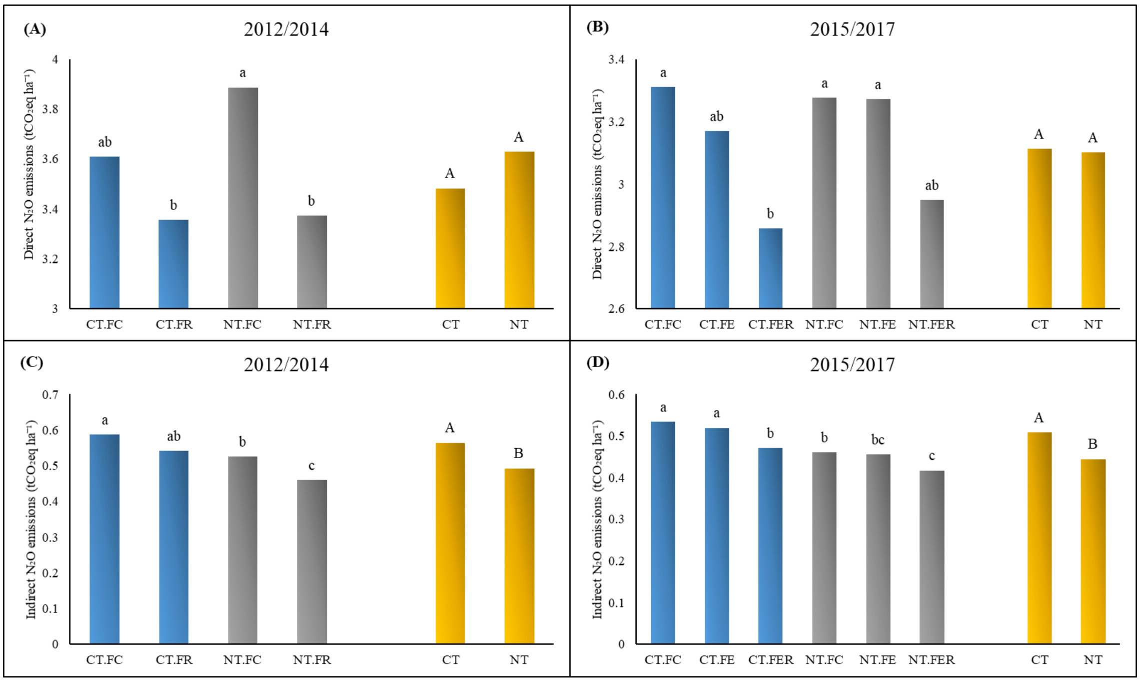

Figure 5.3. Direct (A); (B) and indirect N_2O (C); (D) emissions under CT (conventional tillage) and NT (no-tillage) treatments in 2012/2014 and 2015/2017. Letters indicate statistical difference among different treatments ($P \leq 0.05$). FC, conventional fertiliser NAC 27; FR, reduced fertiliser NAC 27; FE, conventional fertiliser ENTEC 26; FER, reduced fertiliser ENTEC 26.

As it can be observed in 2012/2014 (Figure 5.4. A), the effect of N-fertilization rates did not record significant differences under CT management, while the N_2O emissions from N leaching produced from the use of FC was significantly higher (0.4 tCO_{2eq} ha^{-1}) than the ones produced from FR (0.35 tCO_{2eq} ha^{-1}) under NT system. During this period of the study, the lowest rate of N-fertilization (FR) under NT management resulted in the lowest N_2O emissions from N leaching. In 2015/2017 (Figure 5.4. B), the highest N_2O emissions were observed with the application of FC and FE (0.4 tCO_{2eq} ha^{-1}) while they were the lowest with FER (0.36 tCO_{2eq} ha^{-1}) under CT system. Under NT management, N_2O

emissions from N leaching did not underline significant differences between the different fertilization rates.

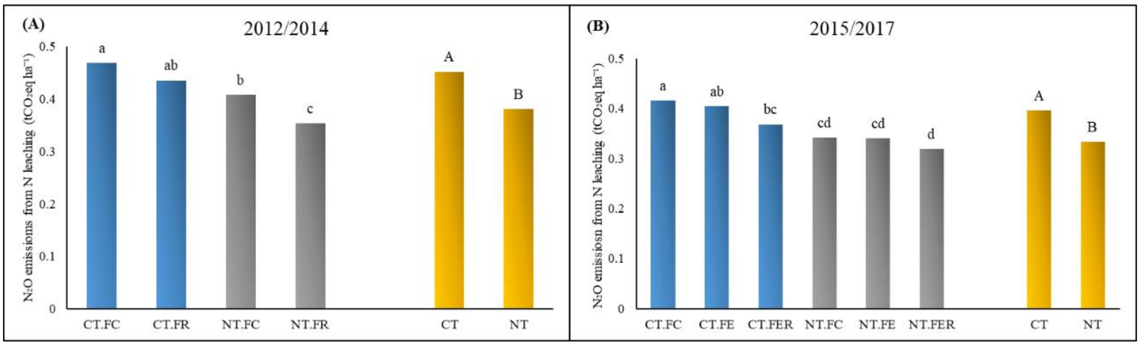

Figure 5.4. N_2O emissions from N leaching into the first 30 cm under CT (conventional tillage) and NT (no-tillage) systems in 2012/2014 (A) and 2015/2017 (B). Letters indicate statistical difference among different treatments ($P \leq 0.05$). FC, conventional fertiliser NAC27; FR, reduced fertiliser NAC 27; FE, conventional fertiliser ENTEC 26; FER, reduced fertiliser ENTEC 26.

5.3.4. Assessment of the maize carbon footprint

According to Figure 5.5 (A), maize grain yield was 4.5% significantly higher in NT plots than in CT plots in 2012/2014. The highest yield was recorded under NT system with FC and FR applications reaching 19.1 and 18.1 t ha^{-1} respectively. While CT plots recorded a quantity of 17.4 t ha^{-1} with FC application and the lowest yield was obtained by the use of FR reaching 16.6 t ha^{-1}. In 2015/2017, the grain yield was, also, significantly higher by 2.5% in NT plots than in CT plots, with the highest amount recorded under NT management and the application of FC and FE reaching 14.3 and 14.4 t ha^{-1} respectively and the lowest was under CT system with FC application (13.2 t ha^{-1}). The results obtained in this study helped the estimation of the maize CFP reflected in Figure 5.5 (C and D). However, it was interesting to evaluate the variation of CFP when including or excluding the SOC changes. When excluding the SOC from the calculation, in 2012/2014 (Figure 5.5. C), the mean CFP was significantly higher under CT management (0.4 tCO_{2eq} t^{-1}) than under NT system (0.3 tCO_{2eq} t^{-1}). It was also the highest with FC under CT system and the lowest with FR under NT treatment. In 2015/2017 (Figure 5.5. D), CT management recorded significantly higher mean CFP (0.5 tCO_{2eq} t^{-1}) than NT system (0.4 tCO_{2eq} t^{-1}). In this case, CT plots with FC displayed higher CFP (0.5 tCO_{2eq} t^{-1}) than FER (0.4 tCO_{2eq} t^{-1}) while NT plots did not record significant differences between the different

N-fertilization rates. Nevertheless, when including the SOC changes in the calculation of the CFP the results obtained completely change to negative values. However, in 2012/2014 the CFP is still significantly higher under CT treatment (-0.16 tCO_{2eq} t^{-1}) than under NT management (-0.20 tCO_{2eq} t^{-1}). The N-fertilization rates did not show significant differences except for FR under NT system, which recorded the lowest CFP (-0.23 tCO_{2eq} t^{-1}). In 2015/2017, the difference remained significantly higher under CT system (-0.23 tCO_{2eq} t^{-1}) than under NT treatment (-0.31 tCO_{2eq} t^{-1}). While, the CFP based on the fertilization rates did not display significant differences under CT treatment, but was significantly higher than the treatments under NT system. Both FC and FE recorded higher CFP reaching -0.30 tCO_{2eq} t^{-1} than FER (-0.35 tCO_{2eq} t^{-1}).

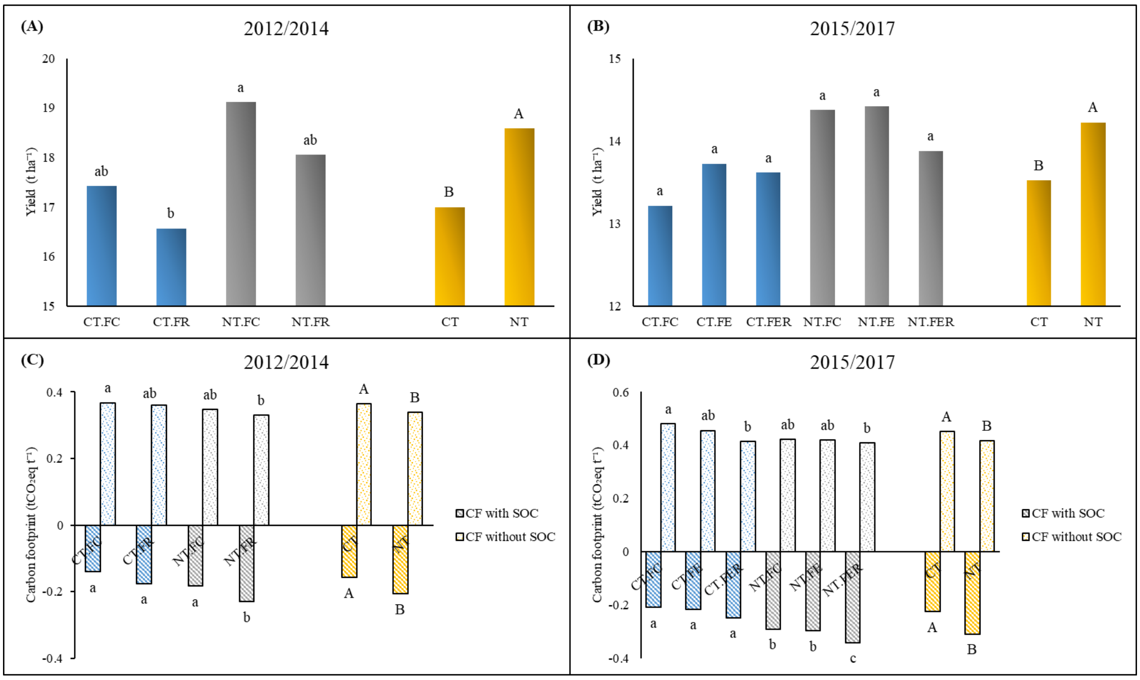

Figure 5.5. Mean grain yield (A) and (B) and mean carbon footprint (C) and (D) under CT (conventional tillage) and NT (no-tillage) managements and N-fertilisation rates in 2012/2014 and 2015/2017. Letters indicate statistical difference among different treatments (*P*≤0.05). FC, conventional fertiliser NAC27; FR, reduced fertiliser NAC27; FE, conventional fertiliser ENTEC26; FER, reduced fertiliser ENTEC26.

In this research, the CFP was obtained using the agricultural inputs such as N fertilizers, which ranged from 47 to 50% under both tillage systems in 2012/2014, from 45 to 48% under CT treatment and from 46 to 49% under NT system in 2015/2017. Irrigation also played an important part in the CFP estimation with percentages of 29 and 31% in 2012/2014 and 2015/2017 respectively, followed by the emissions produced from the fossil fuel combustion that ranged from 8 to 11 % in both periods of the study. Besides, direct and indirect N_2O emissions released after N application accounted for a mean of 53 and 51 % under CT and NT systems respectively in 2012/2014 and ranged from 66 and 67% under NT and CT managements respectively in 2015/2017 and were the main factor responsible of the maize CFP increase.

5.4. Discussion

5.4.1. Energy inputs of the maize production

The energy input of the diesel consumed to assure the different field operations was greater under CT than NT management; this was highly expected because of the ploughing and the weed elimination by cultivator before the crop installation and by fertilization and pesticides application (Table 5.3). Khaledian et al (2010) reported that diesel consumption for different cultivated crops (maize, sorghum (*Sorghum bicolor* L. *Moench*) and wheat (*Triticum turgidum* L. *var durum*) comprised 8-10% of total input in direct seeding into mulch (DSM) compared to 16-17% under CT. These values were almost similar to the ones obtained in our work; we also took into consideration the electricity consumed for irrigation and the energy established for the irrigation equipment, which represented 15 % of the direct energy inputs in both tillage systems and was comprised between 9 and 23 % according to Khaledian et al. (2010).

Indirect energy presented 73-76% of total energy inputs where 49% of these values were attributed to synthetic fertilizers, especially N-fertilizers in accordance with Deike et al. (2008) and Khaledian et al. (2010). For maize production in Italy, Borin et al. (1997) found a value of 48-51% for fertilization being 43-46% in our study. These results illustrate the importance of indirect energy inputs and that maize production depends mostly on fossil energy sources. It is also worth underlining that the total amount of CO_2 that manufacturing fertilizers produce can reach 1.0 t ha^{-1} followed by fuel combustion (0.4 t ha^{-1}) and manufacturing machinery (0.1 t ha^{-1}) (Taki et al. 2012).

The proportion of total energy input accounted for pesticides ranged to almost 1.5% in CT plots compared to 4% for NT plots (Table 5.3). These percentages are lower than the reported range of 2 and 6% found by Khaledian et al. (2010), and the earlier findings by Hülsbergen et al. (2001) that consisted of 3.6 and 5.1% with CT and DSM, respectively. The decrease of the percentage of the energy input of pesticides we faced may be caused by higher input of synthetic fertilizers in comparison with the inputs of the authors cited above. Deike et al. (2008) emphasised that the application of pesticides, which in our study was due to an important proportion of the glyphosate applied in NT system, has a minor contribution in energy input but has considerable importance for energy output to obtain a higher yield.

5.4.2. Effect of tillage systems on SOC changes over the years

The results obtained in our study (Figure 5.1) highlighted the importance of the adoption of NT management throughout the years because of the higher SOC content in the first 30 cm soil layers compared to CT system. This was supported by Varvel and Wilhem (2011), who conducted a study on rainfed maize and found that NT management led to an accumulation of 34.5 t ha^{-1} of SOC in the first 30 cm after 20 years. While different tillage treatments including chisel (28.7 t ha^{-1}), disk (30.8 t ha^{-1}) or plough (30.1 t ha^{-1}) displayed lower SOC values. Follet et al (2012), did not only confirm that maize grown continuously under NT system helped the sequestration of significant amounts of C but also, they reported an increase in soil C in soil layers going up to 150 cm in depth in maize NT plots as compared to CT plots. Carbonell-Bojollo et al. (2015) reported higher SOC under NT treatment that was comprised between 20.5 and 21.4 t ha^{-1} than under CT, which varied from 17.7 to 19.2 t ha^{-1} in the southern soils of Spain. In the North Temperate Zone in China, the data obtained by Zhang et al (2018) showed that NT continuous maize crop planted in 2013, had the highest SOC storage (73.4 t C ha^{-1}) in 30 cm profile of a clay loam soil classified as Typic Hapludoll while CT had the lowest SOC storage (57.6 t C ha^{-1}). Zhang et al (2019), also reported in a study of a continuous maize crop planted in 2016 on a black soil (36 % clay, 24% silt and 40% sand) that no tillage and ridge tillage managements showed a greater accumulation in SOC storage compared to mouldboard plough in the entire ploughed layer (0-20 cm). However, they did not record any difference in SOC storage for the 0-30 cm depth.

The different managements of plant residues left after the harvest, mainly explain the difference between tillage systems recorded in this study. In fact, according to Blanco-Canqui (2013), the removal of residues influences SOC by directly decreasing C inputs and by accelerating decomposition of the remaining SOC. Anderson-Teixeira et al. (2009) found that 10 years of annual maize residue removal reduced SOC storage by almost 8.0 t ha^{-1} for the 30 cm depth. Zhang et al. (2018) have also confirmed that CT with annual residue removal caused SOC to decrease in the different soil layers studied and led to a large decline in SOC storage in 0-30 cm depth of about 9.0 t ha^{-1}. To highlight the importance of crop residues, they found that the SOC storage increased at a rate 0.2-0.8 t ha^{-1} when residues were returned to soil. Adoption of NT practice can increase SOC storage with the expansion of experimental duration; it was assumed that C sequestration could continue at a mean rate of 0.37 t C ha^{-1} per year for 20 years following conversion from CT to NT management. The rate of sequestration was assumed to then decline linearly for another 20 years, with SOC reaching new steady state 40 years after conversion to NT (Lal et al., 1998; West and Marland, 2002).

The interactive effects of the residue at the soil surface under NT management and the absence of soil disturbance contribute to enhance SOC contents at/or near the soil surface. In fact, tillage disturbance is the dominant factor reducing the soil carbon stabilization within micro aggregates (Liu et al., 2016). Huang et al. (2010) reported that tillage treatments altered significantly aggregate distributions, a greater percentage of macro-aggregates and micro-aggregates was found under NT treatment compared to CT system. They declared that NT stimulated soil C accumulation within micro-aggregates, which led to the increase of total SOC by 18.1% compared to CT treatment under long-term maize monoculture in Northeast China.

5.4.3. Effect of tillage systems and N-fertilization rates on N_2O emissions

The N_2O emissions induced from the maize production were basically originated from the application of synthetic fertilizers. Up to 88% were direct emissions produced from the nitrification and denitrification processes that occur in the soil while 11% consisted of indirect emissions from N leaching (Figure 5.2). These results agreed with the meta-analysis conducted by Tongwane et al (2016) in South Africa, where they found that 72% of emissions from application of synthetic fertilizers is from production of cereal crops. 75% of the emissions in crop production from synthetic N fertilizer were direct, 16% were indirect emissions from leaching/runoff of N from the fertilizer and 9% were from atmospheric N deposition. Although there were no significant differences, NT treatment displayed higher direct N_2O emissions than CT treatment in 2012/2014 (Figure 5.2. A), this could be explained by the higher soil moisture that stimulate the nitrification and denitrification processes. According to Groffman (1984), high nitrification activity was observed at 0 to 5 cm in a well-drained sandy clay loam soil under NT management relative to CT system. This was probably caused by greater NH_4^+-N availability, higher pH and higher soil moisture in NT soil at this depth. At lower depths the factor NH_4^+-N availability, pH and moisture interacted to cause higher levels of nitrification activity in CT soil than NT soil. Ploughing and subsequent decomposition and mineralization of residues created an increase of the NH_4^+-N availability at depth in CT soil relative to NT soil. Groffman (1984) also found in his study that denitrification activity measured by the anaerobic assay method was consistently higher in the top 5 cm of NT soil than in CT soil with an opposite pattern observed at lower depths. The absence of statistical difference between tillage systems may be caused by the consideration of the 30 cm of soil for both CT and NT managements and not considering different layers of the soil. Moreover, the application or the presence of crop residues in the soil surface has been shown to increase N_2O emissions (Aulakh et al., 1991). Because NH_4^+, NO^{3-} and organic are used in the nitrification and denitrification processes, the mineralisation of crop residues increases N_2O production. Vigil and Kissel (1991) reported that applying residues with a low C: N ratio encouraged mineralisation, but a high C: N ratio advanced N immobilisation. Baggs et al. (2000) also confirmed that the presence of straw with high C: N ratio on the soil surface may increase the immobilisation of N fertilizers applied and thus decrease the denitrification reactions and N_2O emissions. When no straw or straw with small C: N ratio is present on the soil surface, the N immobilisation probably will not occur and more N will be available for nitrification and denitrification processes and higher N_2O emissions may occur. In this context, it is necessary to mention the effect of the vetch crop residues that were left on the soil surface in 2010 under NT management and incorporated into the soil under CT system. This crop had a relatively low C: N ratio (11:1) which would encourage the mineralisation process in the first 30 cm of the soil in 2012/2014. In fact, N_2O emissions in 2012/2014 were slightly higher, ranging from 3.5 to 3.6 t CO_{2eq} ha^{-1} under CT and NT systems respectively, than the ones produced in 2015/2017 that reached 3.1 t CO_{2eq} ha^{-1} under both tillage systems (Figure 5.2. A and B). This decrease may be explained that during 2015/2017, mainly maize crop residues with high C: N ratio were left or incorporated to the soil leading to a lower N mineralisation and higher N immobilisation. The highest N_2O emissions were attributed to direct emissions in both tillage systems, nevertheless, N_2O emissions from N leaching (Figure 5.3) help emphasise the risk of groundwater contamination or surface water eutrophication that threaten human and animals' health. A previous study conducted in a region of north China showed that the nitrate leaching decreased sharply from 149 to 6 kg N ha^{-1} $year^{-1}$ if the N rate was reduced from 800 to 200 kg N ha^{-1} yr^{-1} (Li et al., 2007). This observation was confirmed by the study of Huang et al. (2011) where they found that decreasing the amount of N from 720 to 360 kg N ha^{-1} $year^{-1}$ reduced the nitrate leaching from 177.8 to 52.5 kg N ha^{-1} $year^{-1}$. In addition, a meta-analysis study of 279 observations on nitrate leaching confirmed that fertilizer management reduced N leaching by 40% if the reduced fertilizer rate was applied to match crop N demand (Quemada et al., 2013). Consequently, the overuse of synthetic fertilizers, higher than the amount of N taken up by the crops, is the key factor responsible for high nitrate leaching in our study.

As expected in our study the high fertilization rates (FC and FE) under both tillage systems led to higher N2O emissions. The high emissions from the synthetic fertilizers applied were obtained as a result of high application rates that were aimed to improve soil fertility and the crop productivity (Tongwane et al., 2016). Besides the high rates of the synthetic fertilization, irrigation tended to promote the nitrification and denitrification by increasing soil moisture, which would enhance N_2O emissions (Li et al., 2012). For indirect emissions, literature indicated that irrigation increased N leaching while reducing NH_3 volatilisation, however direct N_2O emissions were the main pathway (Zhang et al., 2018). Quemada et al. (2013) reaffirmed that management practices that adjust water application to crop needs reduced N leaching by 80%. Zhang et al. (2018) also found that N_2O emissions was greater in irrigated continuous maize crop (0.6%) than in rainfed system (0.4%).

5.4.4. *Assessment of the maize carbon footprint*

The maize CFP depended on a high level on the variation of SOC changes, actually when excluding the SOC, the maize CFP in our study generally ranged from 0.4 to 0.5 tCO_{2eq} t^{-1} under CT management and from 0.3 to 0.4 tCO_{2eq} t^{-1} under NT system for the whole duration of the experiment (Figure 5.5). These results are in consistency with the ones found by Cheng et al. (2014) who reported a CFP value of 0.4 tCO_{2eq} t^{-1} for an irrigated maize crop in an analysis of national statistics data in China. In semiarid conditions of Shanxi, China, Yan et al. (2015) reported lower maize grain yield (6.2 t ha^{-1}) and CFP (0.4 tCO_{2eq} t^{-1}) than our findings. However, comparing with maize crop cultivated in a humid area of China, the CFP ranged from 0.3 to 0.4 tCO_{2eq} t^{-1} which was closer to our results.

When including the SOC changes, the CFP ranged from -0.3 to -0.1 tCO_{2eq} t^{-1} under CT management and from -0.3 to -0.2 tCO_{2eq} t^{-1} under NT system also for all the study years.

The same CFP variation was observed in a study conducted by Zhang et al. (2016) on irrigated summer maize, the CFP switched to negative values when adding the SOC changes under NT system. When not considering SOC changes in the calculation of CFP, a four wheat cropping systems emitted a mean of 0.64 and 0.3 tCO_{2eq} t^{-1} in wet and dry years respectively. However, when soil C changes were included, the maize CFP displayed -0.3 tCO_{2eq} t^{-1} in in wet years and -0.6 tCO_{2eq} t^{-1} in dry years (Gan et al., 2012). In our study, the energy inputs of N-fertilizers as well as direct and indirect N_2O emissions strongly affected the maize CFP; this agrees with the findings of Fumagalli (2015) who reported that emissions from synthetic fertilizers and from direct and indirect N_2O accounted for a mean of 67% of the total emissions. Qi et al. (2018) reaffirmed that N_2O emissions from soil were the main contributors to the CFP of a spring maize, accounting for 36.8 to 54% of the CFP. Followed by the emissions from synthetic fertilizers that ranged from 36.8 to 49.4% of the CFP of maize production at yield-scale, which ranged from 10.7 to 17.0 t ha^{-1}.

Moreover, the value of the CFP for our maize crop was significantly influenced by SOC changes. Including the SOC over the 6-year period in the calculation reversed the CFP values from positive to negative values in both tillage systems, which indicates a net C sink capacity that is significantly higher under NT management than under CT system. GHGs associated with crop production inputs can be offset by greater carbon conversion from atmospheric CO_2 into plant biomass and ultimately sequestered into the soil (Liu at al., 2016). However, it is necessary to mention that the sequestration of C into the soil does not only depend on the soil management but also on the soil texture and its physicochemical properties, the quality and distribution of the crop residue left on the soil surface and the climatic conditions.

5.5. Conclusion

This study assessed the effects of two tillage systems (CT and NT) and different N fertilization rates on the CFP and its components of spring maize production to identify practices that help lowering the agricultural inputs without causing a drastic drop of maize grain yield and reduce the environmental impacts in the area. The data presented in our study showed that total GHG emissions from agricultural inputs ranged from 2.1 to 2.3 t CO_{2eq} ha^{-1} under CT treatment and from 2.1 to 2.2 t CO_{2eq} ha^{-1} under NT treatment. In both periods of the study, diesel and electricity inputs (direct energy) and machinery use were higher under CT treatment while synthetic fertilizers, pesticides and water applications and maize seed (indirect energy) were greater under NT treatment. This difference between tillage systems was caused by the preparatory work before planting under CT and by weeds control under NT system. Besides agricultural inputs, total N_2O emissions affected significantly the CFP of spring maize but there was no significant difference observed between tillage systems. However, N_2O emissions from N leaching were greater in ploughed soil than in no disturbed soil. The application of synthetic fertilizers was the key factor of the increase of the agricultural inputs as well as the N_2O emissions. In fact, reducing the amount of N-fertilization applied decreased the energy inputs by 0.1 tCO_{2eq} ha^{-1} in both periods of the study. In addition, this reduction lowered the total N_2O emissions by 0.3 and 0.6 tCO_{2eq} ha^{-1} under CT and NT system respectively in 2012/2014, and by 0.5 and 0.4 tCO_{2eq} ha^{-1} under CT and NT system respectively in 2015/217. The reduction of N fertilizers had also affected mean grain yield which decreased by 0.9 and 1.0 t ha^{-1} under CT and NT system in 2012/2014 and by 0.1 and 0.5 t ha^{-1} under CT and NT system respectively in 2015/2017.

Moreover, this study highlighted the importance of the SOC changes in quantifying the CFP, it was observed that by excluding these changes the CFP recorded high and positive values under both tillage systems, and by including the SOC, these values lowered significantly to reach negative values. This CFP decrease reflects that the maize crop was acting as net sink for C storage, with NT system providing the greatest and CT system the lowest C sequestration benefit. Therefore, the SOC exclusion or its inclusion are very important to evaluate the magnitude of the CFP of spring maize and to quantify the turnover of the GHG emissions and C sequestration. It is also important to assess the CFP of the crop in order to reduce the production costs and help producers to avoid unnecessary financial wastes. Overall, conversion from CT to NT system and reduction of synthetic fertilization inputs of spring maize in the region of Castile and Leon, could provide potential solutions to mitigate GHGs emissions without causing the grain yield to drop severely.

References

Anderson-Teixeira, K. J., Davis, S.C., Masters, M. D., Delucía, E. H. 2009. Changes in soil organic carbon under biofuel crops. Gcb Bioenergy 1 (1), 75-96.

Audsley, E., Stacey, K., Parsons D. J., Williams, A. G. 2009. Estimation of the greenhouse gas emissions from agricultural pesticide, manufacture and use. Cranfield University.

Aulakh, M.S., Doran, J. W., Walters, D.T., Mosier. A. R., Francis, D. D. 1991. Crop residue type and placement effects on denitrification and mineralization. Soil Science Society of. America Journal 58, 1020-1025.

Baggs, E. M., Rees, R. M., Smith, K, A., Vinten, A. J. A. 2000. Nitrous oxide emission from soil after incorporating crop residues. Soil Use and Management. 16, 82-87.

Blanco-Canquí, H. 2013. Crop residue removal for bioenergy reduces soil carbon pools: how can we offset carbon losses? Bioenergy Res. 6, 358-371.

Borin, M., Menini, C., Sartori, L. 1997. Effects of tillage systems on energy and carbon balance in northeastern Italy. Soil & Tillage Research. 62, 229-236.

Campbell, C. A., Zentner, R. P., Selles, F., Biederbeck, V. O., McConkey, B. G., Lemke, R., Gan, Y.T. 2004. Cropping frequency effects on yield of grain, straw, plant N, N balance and annual production of spring wheat in the semiarid prairie. Can J. Plant Sci. 84, 487-501.

Carbonell-Bojollo, R., González-Sánchez, E. J., Repullo Ruibérriz de Torres, M., Ordóñez-Fernández, R., Domínguez-Gimenez, J., Basch, G. 2015. Soil organic carbon fractions under conventional and no-till management in a long-term study in southern Spain. Soil Research. 53 (2), 113-124.

Cheng, K., Yan, M., Nayak, D., Pan, G. X., Smith, P., Zheng, J. F., Zheng, J. W. 2015. Carbon footprint of crop production in China: An analysis of National Statistics data. J. Agricultural Science 153(3), 422 - 431.

Corsi, S., Friedrich, T., Kassam, A., Pisante, M., Moraes, Sà J. 2012. Soil Organic Carbon Accumulation and Greenhouse Gas Emission Reductions from Conservation Agriculture: A literature review In. Integrated Crop Management vol. 16, 2012.

Deike, S., Pallutt, B., Christen, O. 2008. Investigations on the energy efficiency of organic and integrated farming with specific emphasis on pesticide use intensity. European Journal of Agronomy, vol 28. Issue 3, 461-470.

Erdal, G., Esengun, K., Erdal H., Gunduz O. 2007. Energy use and economical analysis of sugarbeet production in Tokat province of Turkey. International Journal of Energy. 32, 35-41.

Follett, R. F., Vogel, K. P., Varvel, G. E., Mitchell, R. B., Kimble, J. 2012. Soil Carbon Sequestration by Switchgrass and No-Till Maize Grown for Bioenergy. Bioenergy Research. 5(4), 866–875.

Fumagalli, M. 2015. Alternative nitrogen management practices to reduce carbon footprint of maize production. Italian Journal of. Agrometeorology. 20, 21-32.

Gan, Y., Liang, C., Campbell, C. A., Zentnerm, R. P., Lemke, R. L., Wang, H., Yang C. 2012. Carbon footprint of spring wheat in response to fallow frequency and soil carbon changes over 25 years on the semi-arid Canadian prairie. European. Journal of Agronomy. 43, 175-184.

Gan, Y., Liang, C., Chai, Q., Lemke, R. L., Campbell, C. A., Zentnerm, R. P. 2014. Improving farming practices reduces the carbon footprint of spring wheat production. Nature Communications. 5, 5012.

Goss, M.J., Howse, K. R. 1993. Losses of nitrate in water draining from under autumn-sown crops established by direct drilling or mouldboard ploughing. European Journal of Soil Science. 44 (1), 35-48.

Gregorich, E. G., Rochette, P., Vanden Bygaart, A. J., Angers, D .A. 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil & Tillage Research. 83, 52-72.

Groffman, P. M. 1984. Nitrification and denitrification in conventional and no-tillage soils. Soil Science Scociety of America Journal. 49, 329-334.

Hérnanz, J. L. 2007. Eficiencia energética en Agricultura. El caso de la Agricultura de Conservación. Seminario Científico Técnico: Agricultura de Conservación y Ahorro de Energía. Madrid. Marzo 2007.

Huang, S., Sun, Y. N., Rui., W.Y., Liu., W. R. Zhang., W. J. 2010. Long-Term Effect of No-Tillage on Soil Organic Carbon Fractions in a Continuous Maize Cropping System of Northeast China. Pedosphere 20(3), 285-292.

Huang, M., Liang, T., Ou-yang, Z., Wang, L., Zhang, C., Zhou, C. 2011. Leaching losses of nitrate nitrogen and dissolved organic nitrogen from yearly two crop system, wheat-maize, under monsoon situations. Nutrient Cycling in Agroecosystems. 91, 77-89.

Hülsbergen, K. J., Feil, B., Biermann, S., Rathke, G. W., Kalk, W. D., Diepenbrock, W. A. 2001. Method of energy balancing in crop production and its application in a long-term fertilizer trial. Agriculture Ecosystems & Environment. 86, 303-321.

IPCC. 2006. IPCC guidelines for National Greenhouse Gas Inventories. Vol.4: Agricultural, Forestry and Other Land Use. IGES, Kanaggawa, Japan.

IPCC. 2007. Cambio climático 2007: Informe de síntesis. IPCC, Ginebra, Suiza.

ISO TS 14067. 2013. Greenhouse Gases-carbon Footprint of Products-requirements and Guidelines for Quantification and Communication. International Organization for Standardization, Geneva, Switzerland.

Khaledian, M. R., Mailhol, J. C., Ruelle, P., Mubarak, I., Perret S. 2010. The impacts of direct seeding into mulch on the energy balance of crop production system in the SE of France. Soil & Tillage Research. 106(2), 218–226.

Kitani, O. 1999. CIGR Handbook of Agricultural Engineering. Volume V. Energy and biomass engineering. Trans ASAE.

Lal, R. 2004a. Carbon emission from farm operations. Environment International 30(7), 981–990.

Lal, R. 2004b. Soil carbon sequestration impacts on global climate change and food security. Science Vol 304. Issue 5677, 1623-1627.

Lal, R., Negassa, N., Lorenz K. 2015. Carbon sequestration in soil. Current Opinion in Environmental Sustainability. 15, 79-86.

Li, X., Hu, C., Delgado, J. A., Zhang, Y., Ouyang, Z. 2007. Increased nitrogen use efficiency as a key mitigation alternative to reduce nitrate leaching in North China plain. Agricultural Water Management 89, 137-147.

Li, H., Qiu, J. J., Wang, L. G., Xu, M. Y., Liu, Z. Q., Wang, W. 2012. Estimates of N2O emissions and mitigation potential from spring maize field based on DNDC model. Journal of Integrative Agriculture. 12, 2067-2078.

Liu, C., Cutforth, H., Qiang, C., Gan, Y. 2016. Farming tactics to reduce the carbon footprint of crop cultivation in semiarid areas. A review. Agronomy for Sustainable Development. (2016), 36-69.

Ma, B. L., Liang, B.C., Biswas, D. K., Morrison, M. J., McLaughlin, N. B. 2012. The carbon footprint of maize production as affected by nitrogen fertilizer and maize-legume rotations. Nutrient. Cycling in. Agroecosystems. 94, 15-31.

Mughal, A. Q., Amjad, N. 2012. Evaluation of Inputs and Outputs Energy for Maize Grain Yield. Sarhad Journal of Agriculture. 28(3).

Qi, J. Y., Yang, S. T., Xue, J. F., Lui, C. X., Du, T. Q., Hao, J. P., Cui, F. Z. 2018. Response of carbon footprint of spring maize production to cultivation patterns in the Loess Plateau, China. Journal of Clean Production. 187, 525-536.

Quemada, M., Baranski, M., Nobel-de-Lange, M.N.J., Vallejo, A., Cooper, J. M. 2013. Meta-analysis of strategies on control nitrate leaching in irrigated agricultural systems and their effects on crop yield. Agriculture, Ecosystems and Environment. 174, 1-10.

Reay, D. S., Smith, K. A., Edwards, A. C., Hiscock, K. M., Dong, L. F., Nedwell, D. B. 2005. Indirect nitrous oxide emissions: Revised emission factors. Environmental Science. 2, 153–158.

Rochette, P., Worth, D. E., Lemke, R. L., McConkey, B. G. Pennock, D. J. Wagner-Riddle, W., Desjardins, R. L. 2008. Estimation of N2O emissions from agricultural soils in Canada. I. Development of a country-specific methodology. Canadian Journal of Soil Science. 88, 641-654.

Seguin, B., Soussana, J. F. 2008. Emissions de gaz à effet de serre et changement climatique : causes et conséquences observées pour l'agriculture et l'élevage. Courrier de l'environnement de l'INRA, 55, 79-91.

Signor, D., Pellegrino, Cerri C. E. 2013. Nitrous oxide in agricultural soils : a review. Pesquisa Agropecuária Tropical Goiânia v. 43, 3, 322-338.

Smith, P. Martino, D., Cai, Z., Gwary, D., Janzen, H. H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C. W., Scholes, B., Sirotenko, O., Howden, S. M., Mcallister, T.A., Pan, G., Romanenkov, V., Schneider, U. A., Towprayoon, S., Wattenbach, M., Smith, J. 2008. Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. B. 363, 789-813

Taki, M., Ghasemi Mobtaker, H., Monjezi, N. 2012. Energy input-output modelling and economical analyse for corn grain production in Iran. Elixir Agriculture. 52, 11500-11505.

Tongwane, M., Mdlambuzi, T., Moeletsi, M., Tsubo, M., Mliswa, V., Grootboom, L. 2016. Greenhouse gas emissions from different crop production and management practices in South Africa. Environmental Devolpment. 19, 23-35.

United Nations. 2019. https://www.un.org/development/desa/en/news/population/world-population-prospec_ts-2019.html

Varvel, G. E., Wilhelm, W. W. 2011. No-tillage increases soil profile carbon and nitrogen under long-term rainfed cropping systems. Soil & Tillage Research. 114(1), 28–36.

Vigil, M. F., Kissel D. E. 1991. Equations for estimating the amount of nitrogen mineralized from crop residues. Soil Science Scociety of America Journal. 55, 757-761.

West, T. O., Marland, G. 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: Comparing tillage practices in the United States. Agriculture Ecosystems and Environment. 91(1–3), 217–232.

Zhang, X. Q., Pu, C., Zhao, X., Xue, J. F., Zhang, R., Nie, Z. J., Zhang, H. L. 2016. Tillage effects on carbon footprint and ecosystem services of climate regulation in a winter wheat-summer maize cropping system of the North China Plain. Ecological Indicators. 67, 821–829.

Zhang, W., He, X., Zhang, Z., Gong, S., Zhang, Q., Zhang, W., Liu, D., Zou, C., Chen, X. 2018. Carbon footprint assessment for irrigated and rainfed maize (*Zea mays L.*) production of the loess plateau of China. Biosystem Engineering. 167, 75-86.

Zhang, Y., Li, X., Gregorich, E. G., McLaughlin, N.B., Zhang, X., Guo, Y., Liang, A., Fan, R., Sun, B. 2018. No-tillage with continuous maize cropping enhances soil aggregation and carbon storage in Northeast China. Geoderma, 330, 204-2011.

Zhang, Y., Li, X., Gregorich, E.G., McLaughlin, N.B., Zhang, X., Guo, Y., Gao, Y., Liang, A. 2019. Evaluating storage and pool size of soil organic carbon in degraded soils: Tillage effects when crop residue is returned. Soil & Tillage Research. 192, 215-221.

Yan, M., Cheng. K., Luo, T., Pan, G., Rees, R. M. 2015. Carbon footprint of grain crop production in China – based on farm survey data. Journal of Clean Production 104, 130-138.

Yang, X., Gao, W., Zhang, M., Chen, Y., Sui, P. 2014. Reducing agricultural carbon footprint through diversified crop rotation systems in the North China Plain. Journal of Clean Production 76, 131- 139.

Chapter 6. Conclusions

Conclusions

Maize production in semi-arid areas of Spain is mainly limited by the available water as rainfall is very irregular between and throughout the years. In addition to a weakened soil quality as the organic matter content is less than 2%, which implies low fertility and high susceptibility to degradation. For this reason, agricultural practices are becoming more and more conservative. After several years of studying the effect of conventional tillage and no-tillage systems on different aspects of the maize production in semiarid conditions in Castile and Leon, it comes to the following conclusions:

- ✓ Soil bulk density and soil pH are likely to be affected by the sampling dates, climatic conditions, soil texture, crop type and the traffic of mechanized equipment rather than tillage system; however, the accumulation of crop residues also play a major part in the acidification of the soil (the soil pH was lower in 2017 than in 2011and in the upper layers of the soil profile).
- ✓ The combination of NT practice and crop residue retention on the soil surface is shown to increase the soil organic matter and soil organic carbon especially in the first 30 cm of the soil profile, in addition of a high total nitrogen content compared to CT system even though no significant differences were observed.
- ✓ The soil phosphorus is not particularly a mobile nutrient and its availability in the soil solution for the plant uptake is conditioned by various factors such as compaction, high moisture level, temperature and soil pH. Actually, highly alkaline soils as observed under NT system in this study can cause the P to become tied up in an insoluble phosphate and can create temporary phosphorus deficiency.
- ✓ In this study nitrate leaching was mostly affected by the amount of available water, as the higher concentrations were observed in 2015 (year with the highest irrigation amount), but also could be caused by the high rates of N fertilization as the samples collected for the analysis came from plots with high N fertilizers application
- ✓ The assessment of the soil water content resulted in high mean cumulative SWC of the 3 studied years under NT system than CT system. The years with high water inputs (2015 and 2016) presented high moisture level all along the soil profile resulting in an increase of the water loss by percolation especially in August and September of both years. The last year of the study was characterized by the

irregularity and scarcity of irrigation applications, high temperature and long period of drought resulted in higher SWC in the non-tilled soils especially up to 100 cm soil depth than in the tilled soils. Nevertheless, during the studied months of this year, no water percolation was observed due to higher evapotranspiration and the elevated hydric needs of the maize crop.

- ✓ The maize grain yield and its components were determined and the results obtained demonstrate that the 1000 grain weight had the strongest effect on maize yield followed by the grain number per ear.
- ✓ The harvested grain yield was significantly higher under NT system than CT in 2017 but no significant differences were observed for the rest of the studied years.
- ✓ The cumulative CO_2 flux in the first 48 hours after the mouldboard and cultivator pass was significantly higher in the tilled plots than in the non-tilled plots during the studied years. The monitoring of the phenological stages of the maize crop indicated that the tillage system significantly affected CO_2 emissions during the growing stages of the crop and when soil temperature and moisture due to irrigation increase.
- ✓ The evaluation of the carbon footprint showed that diesel and electricity inputs (direct energy) and machinery use were higher under CT system while synthetic fertilizers, pesticides and water applications and maize seed (indirect energy) were greater under NT treatment. This difference between tillage systems was caused by the preparatory work before planting under CT and by weeds control under NT system. Besides agricultural inputs, total N_2O emissions affected significantly the CFP of spring maize but there was no significant difference observed between tillage systems. However, N_2O emissions from N leaching were greater in ploughed soil than in no disturbed soil.
- ✓ The soil organic carbon changes play a major part in the quantification of the carbon footprint; including the SOC changes in the calculation lowers the CFP and promotes the C sequestration while when excluding it from the calculation, the CFP displays high values which reflects a high agricultural inputs and high emissions of CO_2 from the maize production.
- ✓ In this study, the maize crop acted as a net sink for carbon storage under both tillage system. Yet, the SOC content was significantly higher under NT system

than CT in the first 30 cm of the soil profile which resulted in significantly lower CFP under NT system.

In addition of the benefits cited above provided by the no-tillage system, it also consists of a helpful solution for farmers to lower the increasing costs of machinery, synthetic fertilization and minimize the life cycle of the inputs that are necessary for a decent crop production on a long-term.

Conclusiones

La producción del maíz en zonas semiáridas de España viene principalmente limitada por las precipitaciones que son muy irregulares entre años y a lo largo de los mismos. Además de los suelos que son habitualmente muy pobres en materia orgánica (menor del 2%) lo que implica una baja fertilidad y una alta susceptibilidad a la degradación del mismo. Por esta razón, las prácticas de agricultura se hacen cada vez más conservativas. Tras varios años de estudio del efecto del sistema laboreo convencional y no laboreo sobre diferentes aspectos de la producción del cultivo maíz en condiciones semiáridas como es el caso en Castilla y León, se llega a las siguientes conclusiones:

- ✓ Durante todo el periodo del estudio, el sistema de laboreo generalmente no afecto a la densidad aparente y el pH del suelo, ya que estos parámetros son principalmente afectados por las fechas de muestreo, las condiciones climáticas, la textura del suelo, el tipo de cultivo y el tráfico de equipos mecanizados. Sin embargo, la acumulación de residuos de cultivos sobre la superficie del suelo juega también un papel importante en la acidificación del suelo (el pH del suelo fue menor en 2017 que en 2011 y en las capas superiores del perfil del suelo).
- ✓ Se ha demostrado que la combinación del no laboreo y el hecho de dejar los residuos del cultivo sobre la superficie del suelo aumenta la materia orgánica y el carbono orgánico del suelo, especialmente en los primeros 30 cm del perfil del suelo, además de un mayor contenido de nitrógeno total en comparación con el sistema CT.
- ✓ El fósforo del suelo no es particularmente un nutriente móvil y su disponibilidad en la solución del suelo para la absorción de la planta está condicionada por varios factores como la compactación, el alto nivel de humedad, la temperatura y el pH del suelo. De hecho, los suelos altamente alcalinos, como los observados en el sistema de no laboreo en este estudio, pueden hacer que el P se convierta en un fosfato insoluble y crear una deficiencia temporal de fósforo.
- ✓ En este estudio la lixiviación de nitratos se vio afectada principalmente por la cantidad de agua disponible, ya que las mayores concentraciones se observaron en 2015 (año con la mayor cantidad de riego), pero también podría ser causada por las altas cantidades de fertilización nitrogenada, ya que las muestras recogidas para el análisis provenían de parcelas con alta aplicación de fertilizantes.

- ✓ La evaluación del contenido de agua del suelo dio como resultado una media de agua acumulada alta en los 3 años estudiados bajo el sistema de no laboreo que en el sistema de laboreo convencional. Los años con altos aportes hídricos (2015 y 2016) presentaron un alto nivel de humedad a lo largo de todo el perfil del suelo, lo que resultó en un aumento de la pérdida de agua por percolación, especialmente en agosto y septiembre de ambos años. El último año del estudio se caracterizó por la irregularidad y escasez de las aplicaciones de riego, las altas temperaturas y el largo periodo de sequía, lo que dio lugar a una mayor cantidad de agua acumulada en los suelos no labrados, especialmente hasta los 100 cm de profundidad, que en los suelos labrados. Sin embargo, durante los meses estudiados de este año, no se observó percolación de agua debido a la mayor evapotranspiración y a las elevadas necesidades hídricas del cultivo de maíz.
- ✓ En este estudio se determinó el rendimiento de grano de maíz y sus componentes y los resultados obtenidos demuestran que el peso de 1000 granos tuvo el mayor efecto sobre el rendimiento de maíz, seguido del número de granos por mazorca. La estimación del rendimiento en grano mediante sus diferentes componentes no se vio afectada por los sistemas de laboreo.
- ✓ El rendimiento de grano cosechado fue significativamente mayor en el sistema de no laboreo que en sistema de laboreo convencional en 2017, pero no se observaron diferencias significativas para el resto de los años estudiados.
- ✓ El flujo acumulativo de CO_2 en las primeras 48 horas después del pase de vertedera y el cultivador fue significativamente mayor en las parcelas de laboreo convencional que en las de no laboreo en las campañas estudiadas. Los resultados en el seguimiento fenológico del cultivo de maíz en regadío indicaron que el sistema de laboreo influyó de forma significativa en las emisiones de CO_2 durante los estados vegetativos cuando el cultivo comienza a crecer de forma considerable y la temperatura del suelo y la humedad debida al riego aumentan.
- ✓ La evaluación de la huella de carbono mostró que los insumos de gasóleo y electricidad (energía directa) y el uso de maquinaria fueron mayores en el sistema de laboreo convencional, mientras que el uso de los fertilizantes sintéticos, los pesticidas y las aplicaciones de agua y las semillas de maíz (energía indirecta) fueron mayores en el sistema de no laboreo. Esta diferencia entre los sistemas de laboreo se debió a la preparación del lecho de siembra en el sistema CT y al control

de las malas hierbas en el sistema de no laboro. Además de los insumos agrícolas, las emisiones totales del óxido nitroso afectaron significativamente a la huella de carbono del cultivo de maíz, pero no se observaron diferencias significativas entre los sistemas de laboreo. Sin embargo, las emisiones de N_2O procedentes de la lixiviación del N fueron mayores en el suelo arado que en el suelo no perturbado.

- ✓ Los cambios en el carbono orgánico del suelo juegan un papel importante en la cuantificación de la huella de carbono; su inclusión en el cálculo disminuye la huella de carbono y promueve el secuestro de C, mientras que cuando se excluye del cálculo, la huella de carbono muestra valores elevados que reflejan un alto nivel de insumos agrícolas y altas emisiones de CO_2 de la producción de maíz.
- ✓ En este estudio, el cultivo de maíz actuó como un sumidero neto de almacenamiento de carbono bajo ambos sistemas de laboreo. Sin embargo, el contenido del carbono orgánico del suelo fue significativamente mayor en el sistema de no laboreo que en el sistema de laboreo convencional en los primeros 30 cm del perfil del suelo, lo que resultó en una huella de carbono significativamente menor bajo el sistema de no laboreo.

Además de los beneficios proporcionados por el sistema de no laboreo, esta práctica puede proponer una solución útil para que los agricultores reduzcan los costos crecientes de la maquinaria, la fertilización sintética y minimicen el ciclo de vida de los insumos necesarios para una producción de cultivos más sostenible a largo plazo.