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# SOIL CARBON DYNAMICS AND GREENHOUSE GAS EMISSIONS IN CONSERVATION TILLAGE SYSTEMS AT MULTIPLE SCALES

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SOIL CARBON DYNAMICS AND GREENHOUSE GAS EMISSIONS IN CONSERVATION TILLAGE SYSTEMS AT MULTIPLE SCALES

DISSERTATION

A dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy in the College of Agriculture, Food and Environment at the University of Kentucky

By

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2020


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# ABSTRACT OF DISSERTATION

## SOIL CARBON DYNAMICS AND GREENHOUSE GAS EMISSIONS IN CONSERVATION TILLAGE SYSTEMS AT MULTIPLE SCALES

Conservation tillage practices like no-tillage and reduced tillage have been widely implemented worldwide, with expectations they would provide multiple benefits (e.g., yield enhancement and soil carbon sequestration) for food security and climate adaptation and mitigation. However, the adoption of conservation tillage faces both opportunities and challenges. A knowledge gap still exists regarding the effects of conservation tillage on the carbon cycle in agroecosystems. This dissertation reflects a comprehensive evaluation of conservation tillage at multiple scales using an integrated systems approach, a combination of data synthesis, the agriculture ecosystem model, and field observations and measurements. I first conducted a meta-analysis to assess the effects of no-tillage (one widespread conservation tillage) on crop yield, greenhouse gas (i.e., $CO_2$, $CH_4$, and $N_2O$) emissions, and the global warming potential of major cereal cropping systems in the world. Compared to conventional tillage, no-tillage reduced greenhouse gas emissions and increased crop yield in dry climate conditions. It reduced the global warming potential at sites with acidic soils. Considering the crucial role of soil organic carbon in providing ecosystem services, I further analyzed conservation tillage effects on soil carbon sequestration and the environmental controlling factors. Based on the meta-analysis review, I developed a conceptual tillage module accordingly and integrated it into a process-based agroecosystem model, the DLEM-Ag. At a long-term tillage experiment site in Lexington, KY, the improved model captured the changes and trends in soil organic carbon under different tillage treatments during 1970-2018, with no-tillage retaining more soil carbon than moldboard plow. Model factorial analyses revealed that this was mainly due to the lower $CO_2$ emissions in no-tillage than in the moldboard plow treatments. Then, I expanded the simulation to the maize and soybean croplands in Kentucky to explore the conservation tillage effects on greenhouse gas emissions at the regional scale. Sensitivity analyses showed that, compared to conventional tillage, no-tillage significantly reduced $CO_2$ and $N_2O$ emissions in both croplands. Lastly, the effects of conservation tillage on the coupled carbon and water cycles at the Ohio River Basin were examined using the improved DLEM-Ag model. Simulation results suggested higher crop water productivity in maize and soybean croplands under conservation tillage than under conventional tillage at the basin level. This dissertation is based on and adapted from three articles recently published in peer-review journals and two manuscripts prepared for publication.

KEYWORDS: Conservation Tillage, Greenhouse Gas Emissions, Meta-analysis, Process-based Modeling, Soil Carbon

Yawen Huang

08/06/2020

SOIL CARBON DYNAMICS AND GREENHOUSE GAS EMISSIONS IN CONSERVATION TILLAGE SYSTEMS AT MULTIPLE SCALES

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DEDICATION

To my wife Mengxi Zhai, for her understanding, encouragement, and endless love.

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# CHAPTER 1. GENERAL INTRODUCTION

The expanding global population, escalating food demands, and changing climate are putting tremendous pressure on agricultural production (Reicosky, 2015). Meanwhile, agriculture is one of the major sources of greenhouse gas (GHG) emissions, accounting for 12% of total anthropogenic GHG emissions, and non-point source pollution (IPCC, 2014). The challenge of modern farming in response to climate change is to simultaneously improve crop yield and minimize environmental pollution, making the agriculture system resilient to climate risks and mitigating climate change where possible. It is, therefore, vitally important to develop strategies for sustainably improving agriculture production.

Among all the agriculture technologies and strategies, soil management plays a fundamental role in ensuring the sustainability of agriculture and the future of humankind (Powlson et al., 2011), as keeping our soil healthy and productive is of paramount importance in agriculture. Historically, tillage has been performed because of many benefits, including soil preparation for planting and cultivation, facilitating crop establishment, weed control, and incorporation of manure or fertilizer spread on the soil surface. However, by loosening soil and easing its transport by wind and water, soil erosion and degradation have been major concerns associated with the tillage practice. It has been suggested that we are losing soil faster than nature can make it (Montgomery, 2007).

Conservation tillage, the use of less intensive tillage methods than conventional tillage (CT), was initially promoted as soil and water conservation technology. Conservation tillage is defined as any tillage system that decreases the degree and frequency of tillage passes and maintains at least 30 percent residue cover at planting (CTIC, 2020). In the past several decades, conservation tillage systems, such as no-tillage

(NT) (the absence of mechanical disturbance), reduced tillage (RT), strip tillage, and mulch tillage, have been widely adopted worldwide. Globally, more than 155 million ha of cultivated land is currently under conservation tillage management (Kassam et al., 2014). In the US, conservation tillage accounts for 67% of total US acreage in wheat (2017), 65% in corn (2016), 70% in soybeans (2012), and 40% in cotton (2015; Claassen et al., 2018).

The implementation of conservation tillage can lead to responses of soil properties that differ from CT, thus affecting ecosystem services. While some advantages of conservation tillage are clear (e.g., soil erosion control and less fuel consumption), other effects can be varied (e.g., crop yield (Pittlow et al., 2015; Sun et al., 2020), soil carbon sequestration (Baker et al., 2007; Powlson et al., 2014; Bai et al., 2019; Ogle et al., 2019), GHG emissions (Huang et al., 2018), nutrient leaching (Daryanto et al., 2017), and water use (Skaalsveen et al., 2019)). Climate conditions, soil textural class, and duration of conservation tillage management are the main factors that affect conservation tillage effects on soil physical and chemical properties (Holland 2004; Busari et al., 2015; Sun et al., 2020; Blanco-Canqui and Ruis, 2018). The companion agronomic practices, such as fertilizer use and cover crop use, also contribute to the inconsistent effects (Munawar et al.,1990; Petersen et al., 2011; Cook and Trlica, 2016). Nevertheless, one consensus regarding conservation tillage effects has been that it enhances surface water interception due to surface residues cover, which intercepts rainfall and reduces evaporation (Morris et al., 2010; Busari et al., 2015). It is likely that conservation tillage enhances soil hydraulic properties and plant available water through improving soil structural quality and soil carbon content (Blanco-Canqui and Ruis, 2018). However, conservation tillage does not always improve plant available water compared to CT. Fine-textured soils with high initial organic carbon content can respond slower to NT management than coarse-textured soils with low organic carbon in improving plant available water (Blanco-Canqui and Ruis, 2018).

Changes in soil properties can ultimately affect soil organic carbon (SOC) dynamics, crop growth, and soil GHG emissions. Many have reviewed the effects of tillage on soil carbon (Six et al., 2002; Luo et al., 2010; Powlson et al., 2014; Haddaway et al., 2017; Bai et al., 2019; Ogle et al., 2019). These reviews and meta-analyses have shown beneficial and null effects on SOC due to NT relative to conventional tillage. Compared to CT, conservation tillage reduces soil disturbance and the soil organic matter decomposition rate (Salinas-Garcia and Matocha, 1997) and promotes fungal and earthworm biomass (Lavelle et al., 1999; Brione and Schmidt, 2017), thereby improving SOC stabilization (Liang and Balser, 2012). However, there is a debate on whether conservation tillage practices can enhance SOC stocks or just alter SOC distribution in the soil profile (Baker et al., 2007; Powlson et al., 2014; Bai et al., 2019). It is suggested that crop carbon input differences largely determine the tillage effects on SOC stocks and *vice versa* (Virto et al., 2012). If carbon input increases due to tillage management, SOC stocks are more likely to increase.

The positive productivity responses to NT often occur for rainfed crops in dry climates (Pittelkow et al., 2015; Huang et al., 2018) and moderate- to well-drained soils (DeFelice et al., 2006; Triplett and Dick., 2008). However, in more mesic climates or poorly drained soils, the responses can be more varied. Soil water conservation and retention can be a benefit of NT management under water-limited conditions (Farooq et al., 2011). In contrast, the potential for soil waterlogging, delayed soil warming in spring, and soil compaction can be detrimental to crop growth (Licht and Al-Kaisi, 2005; Ogle et al., 2012). Some studies have argued that, despite the unstable yield responses during the initial years of adopting NT, the long-term use of NT modifies the soil properties in a way (e.g., better soil aggregation and structure, increased SOC storage, and improved soil water availability) that ensures the consistent positive effects of NT (Sindelar et al., 2015; Blanco-Canqui and Ruis, 2018; Cusser et al., 2020).

The impacts of conservation tillage on soil GHG (i.e., $CO_2$, $CH_4$, and $N_2O$) emissions also varies among studies (van Kessel et al., 2013; Zhao et al., 2016; Huang et al., 2018). For example, NT can decrease (Li et al., 2011; Drury et al., 2012; Lu et al., 2016), increase or not affect (Oorts et al., 2007; Yao et al., 2013; Zhang et al., 2016b) soil GHG emissions. Mechanisms responsible for reduced soil $CO_2$ emissions under conservation tillage include: 1) less soil disturbance that keeps SOC from oxidation (Rastogi et al., 2002); 2) improved soil aggregate stability that protects SOC from microbial attack (Abdalla et al., 2013); 3) a lower soil temperature (He et al., 2011; Lu et al., 2016). In contrast, the often-higher soil water availability with conservation tillage can also enhance the microbial activity, thus increasing $CO_2$ emissions (Plaza-Bonilla et al., 2014b). Tillage practices affect $CH_4$ fluxes by altering the soil structure, aeration, hydrological properties, microbial activities, and crop growth (Cai et al., 2003; Mangalassery et al., 2014). Higher $CH_4$ emissions associated with conservation tillage can be attributed to a greater abundance of organic substrates and coincident formation of anaerobic microsites (Zhang et al., 2015). Lower $CH_4$ emissions under conservation tillage might be due to improved soil porosity and gas diffusivity, which facilitates the transport of $CH_4$ to methanotrophs that consume $CH_4$ (Ball et al., 1997; Prajapati and Jacinthe, 2014). Soils under conservation tillage are often wetter and denser compared to CT soils, which potentially favors denitrification and, consequently, higher $N_2O$ emissions (Smith et al., 2001; Ma et al., 2013; Sheehy et al., 2013). However, NT soils may have a slower N mineralization rate than CT soils, leading to less $NH_4^+$ and $NO_3^-$ substrate in NT soils and, therefore, lower $N_2O$ production (Dick et al., 2008; Almaraz et al., 2009a). There is a tendency that the long-term use of NT substantially modifies soil physical properties with improved soil aggregation and aeration status, leading to a reduction in $N_2O$ emissions because of more completed denitrification (Six et al., 2004; van Kessel et al., 2013).

These contrasting effects of conservation tillage on SOC, crop yield, and GHG emissions warrant more efforts to compare how conservation tillage can effectively contribute to food security, climate change adaptation, and mitigation by increasing crop yield and decreasing GHG emissions. Field experiments, especially long-term ones, represent an invaluable approach to study the effects of management practices on agroecosystem variables with high credibility. However, findings from individual experiments are often confined to specific environmental conditions and management factors. Meta-analysis reviews are a useful tool to statistically test the linear relationship between site-specific conditions and the target variables. In addition, simulation tools such as the process-based model provide an opportunity to evaluate the effects of crop management practices at multiple scales, which can help illustrate the spatiotemporal heterogeneity associated with the practices' effects.

This dissertation research generally followed a "top-down" approach. The main objective was to comprehensively assess the effects of conservation tillage on SOC, crop yield, and soil GHG emissions at multiple scales with an integrated systems approach: a combination of data synthesis, the agriculture ecosystem model, and field observations and measurements. This research first conducted two meta-analysis reviews to summarize and quantify the effects of conservation tillage on the major agroecosystem variables, as well as the statistical relationship between these effects and environmental and management factors, based on data collected from worldwide field experiments. Chapter 2 focuses on NT effects on crop yield and GHG emissions and conservation tillage effects on SOC. Then, to further examine the conservation tillage effects at local and regional scales, this research improved an agroecosystem model by implementing a tillage module according to the concept developed from the meta-analysis reviews. Chapter 3 introduces the improved model and uses it to investigate NT effects on soil carbon dynamics in a continuous maize system in well-drained soils with a humid climate. Lastly, Chapters 4 and 5 discuss the regional responses of crop yield, GHG emissions, and crop water productivity to conservation tillage using model simulations.

# CHAPTER 2. GREENHOUSE GAS EMISSIONS, CROP YIELD, AND SOIL ORGANIC CARBON IN CONSERVATION TILLAGE SYSTEMS[1]

## 2.1 Abstract

No-tillage (NT) has been touted as one of several climate-smart agriculture (CSA) management practices that improve food security and enhance agroecosystem resilience to climate change. However, the sustainable effectiveness of NT greatly depends on trade-offs between NT-induced changes in crop yield and greenhouse gas (GHG, i.e., $CH_4$, $CO_2$, and $N_2O$) emissions. Such trade-offs are regulated by climate fluctuations and heterogeneous soil conditions and have not been well addressed. Supporting CSA management decisions requires advancing our understanding of how NT affects crop yield and GHG emissions in different agroecological regions. In this study, a meta-analysis was conducted using 740 paired measurements from 90 peer-reviewed articles to assess the effects of NT on crop yield, GHG emissions, and the global warming potential (GWP) of major cereal cropping systems. Compared to conventional tillage (CT), NT reduced GHG emissions and increased crop yield in dry, but not humid, climates, and reduced the GWP at sites with acidic soils. Across different cropping systems, NT enhanced barley yield by 49%, particularly in dry climates, and decreased the GWP of rice fields through a 22% reduction in both $CO_2$ and $CH_4$ emissions. Our synthesis suggests that NT is an effective CSA management practice because of its potential for climate change mitigation and crop

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[1] Based on Huang, Yawen et al. (2018) “Greenhouse gas emissions and crop yield in no-tillage systems: A meta-analysis” *Agriculture, Ecosystems and Environment* 268, 144-153, and Bai, X.; Huang, Yawen, et al. (2019) “Responses of soil carbon sequestration to climate-smart agriculture practices: A meta-analysis” *Global Change Biology* 25, 2591-2606.

yield improvement. However, the net effect of NT (relative to CT) was influenced by several environmental and agronomic factors (climatic conditions, tillage duration, soil texture, pH, crop species). Therefore, an agroecological setting must be taken into consideration when conducting a comparative evaluation of different tillage practices.

Conservation tillage has been widely adopted to enhance soil organic carbon (SOC) sequestration and to reduce greenhouse gas emissions while ensuring crop productivity. However, current measurements regarding the influences of conservation tillage on SOC sequestration diverge widely, making it difficult to derive conclusions about individual and combined CSA management effects and bringing large uncertainties in quantifying the potential of the agricultural sector to mitigate climate change. We conducted a meta-analysis of 3,049 paired measurements from 417 peer-reviewed articles to examine the effects of three common CSA management practices on SOC sequestration as well as the environmental controlling factors. We found that, on average, biochar applications represented the most effective approach for increasing SOC content (39%), followed by cover crops (6%) and conservation tillage (5%). Further analysis suggested that the effects of CSA management practices were more pronounced in areas with relatively warmer climates or lower nitrogen fertilizer inputs. Our meta-analysis demonstrated that, through adopting CSA practices, cropland could be an improved carbon sink. We also highlight the importance of considering local environmental factors (e.g., climate and soil conditions and their combination with other management practices) in identifying appropriate CSA practices for mitigating greenhouse gas emissions while ensuring crop productivity.

## 2.2 Introduction

**<u>Greenhouse gas emissions and crop yield</u>:** Among all anthropogenic sources, agriculture is estimated to be responsible for 12% of total greenhouse gas (GHG) emissions (IPCC, 2014), particularly global anthropogenic $CH_4$ (39%) and $N_2O$ (76%) emissions (FAO, 2014; WRI, 2014). With increasing demands on agriculture to feed a growing world population, GHG emissions from agroecosystems will likely continue to rise. Climate-smart agriculture (CSA) focuses on methods to maintain or increase food production while simultaneously reducing agriculture's GHG emissions and other environmental side effects under various climate scenarios (FAO, 2013). No-tillage (NT) management has been proposed as a component of CSA (Lipper et al., 2014). In NT, crop residues are left on the soil surface, and only in-row soil is disturbed during seeding (Dinnes, 2004). Compared to conventional tillage (CT), NT exhibits greater potential for soil carbon sequestration, soil quality improvement, and sustained crop productivity (Lal et al., 2007; Lal, 2015; Abdalla et al., 2016). No-tillage was practiced on approximately 111 million ha worldwide in 2009 (Derpsch et al., 2010), and this number reached 155 million ha in 2014 (FAO, 2014). One may reasonably speculate that NT management can potentially have global-scale impacts on the magnitude and spatial patterns of soil GHG emissions and crop production (Figure 2.1). However, from a sustainability perspective, the net effect of NT greatly depends on the trade-offs between NT-induced changes in crop yield and GHG emissions. These trade-offs are regulated by a suite of climatic and soil factors, which have not been well addressed in past studies.

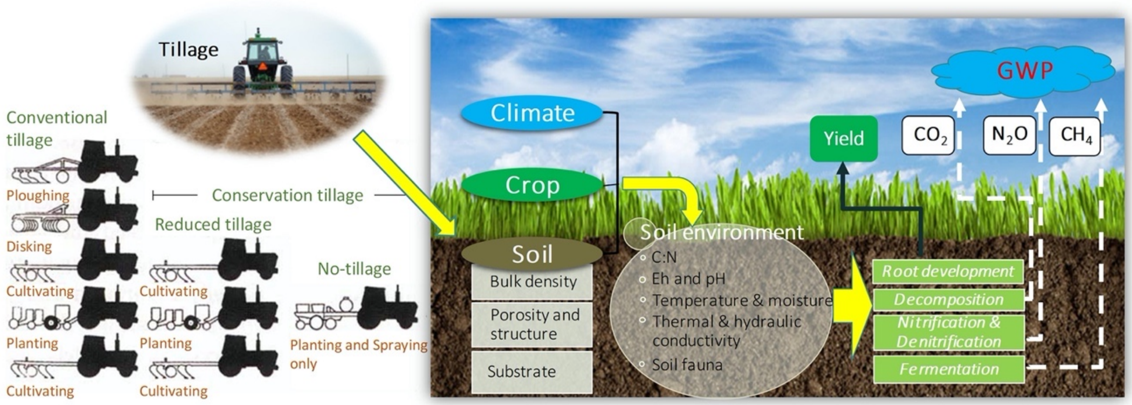


Figure 2.1 Conceptual framework of diverse tillage practices impact on soil processes (biophysical, biophysiological, and biogeochemical), greenhouse gas (GHG) emissions, and crop yields

The precise effects of NT on soil GHG emissions remain controversial and greatly vary among past studies (van Kessel et al., 2013; Zhao et al., 2016). Some studies showed a substantial decrease in soil $CO_2$, $CH_4$, and $N_2O$ emissions with NT (e.g., Li et al., 2011; Drury et al., 2012; Lu et al., 2016), while others reported a significant increase or no difference (e.g., Oorts et al., 2007; Yao et al., 2013; Zhang et al., 2016b). For example, a long-term study in a Mediterranean dryland agroecosystem exhibited a 50% increase in $CO_2$ emission but no difference in $N_2O$ emission in NT compared to CT (Plaza-Bonilla et al., 2014a, b). Kim et al. (2016) reported that total $CH_4$ flux from NT rice fields decreased by 20–27% in the first and second years after NT imposition, but was approximately 36% higher than that from CT fields by the fifth year. Zhang et al. (2016a) also observed a substantial decline in $CH_4$ and $CO_2$ emissions from NT rice fields compared to CT fields. Another rice field experiment exhibited significant $CO_2$ emission reduction but increased $N_2O$ emission in NT (Fangueiro et al., 2017). Therefore, the climate change mitigation efficacy of NT is still uncertain.

Several hypotheses have been proposed to explain the different soil GHG emission responses due to NT. For example, a decrease in soil $CO_2$ emission in NT might be due to carbon protection associated with enhanced soil aggregation and decreased soil temperature (He et al., 2011; Lu et al., 2016), while an acceleration in soil $CO_2$ emission might be due to enhanced microbial activity caused by greater soil moisture availability (Plaza-Bonilla et al., 2014b). Elevated $CH_4$ emission could be attributed to greater abundance of organic substrates and coincident formation of anaerobic microsites (Zhang et al., 2015). Reduced $CH_4$ emission might be associated with improved soil porosity and gas diffusivity, facilitating the transport of $CH_4$ to methanotrophs (Ball et al., 1997; Prajapati and Jacinthe, 2014). NT-induced increases in soil carbon and water content (and therefore higher water-filled pore space) could favor denitrification, ultimately resulting in elevated soil $N_2O$ emission (Ma et al., 2013; Sheehy et al., 2013). In contrast, factors that may contribute to decreased $N_2O$ emission include improved soil structure, lower soil temperature, a limited pool of decomposable organic carbon and low availability of mineral nitrogen due to a slow rate of soil organic matter (SOM) mineralization (Grandy et al., 2006; Chatskikh and Olesen, 2007; Ruan and Robertson, 2013).

With regard to the response of crop productivity to NT, there is also little consensus from the literature (FAO, 2011; Pittelkow et al., 2015). Some studies concluded that crop yield in water-limited conditions often increases with NT adoption (Farooq et al., 2011; Rusinamhodzi et al., 2011). Other studies reported decreased crop productivity in NT due to cooler soil temperatures, soil compaction, and altered soil fertility requirements (e.g., micronutrient deficiencies; Ogle et al., 2012). These contradictory reports suggest that NT effects may be regulated by many variables, including environment (e.g., climate and soil properties) and management (e.g., crop type, fertilization, tillage duration) factors (Daryanto et al., 2017a). These factors may determine the extent to which NT affects the soil carbon and nitrogen cycles and, consequently, soil GHG emissions and crop productivity. Climate, in particular, influences the frequency and amount of precipitation, soil moisture regime, and the production of soil GHGs. Several recent meta-analyses have synthesized data on GHG emissions and crop yield, but these syntheses focused largely on either one GHG species, a specific cropping system, or a specific geographical region. For example, Abdalla et al. (2016) examined $CO_2$ emission in response to NT, and van Kessel et al. (2013) emphasized the central tendency of $N_2O$ emission and the decreasing crop yield trend in NT. While Zhao et al. (2016) assessed $N_2O$ and $CH_4$ emissions in response to NT, their study was restricted to China. Pittelkow et al. (2015) evaluated the influence of crop species and environmental variables on NT and CT crop yields, but they did not account for soil GHG emissions. Assessing the efficacy of NT as a CSA practice requires a simultaneous examination of NT impacts on crop productivity and GHG emissions across different crops and climatic regions.

In light of interest in NT as a CSA practice, and uncertainties associated with NT impacts, we conducted a meta-analysis to simultaneously evaluate NT effects on soil GHG (i.e., $CH_4$, $CO_2$, and $N_2O$) emissions and crop yield. Specifically, we focused on four major cereal crops (barley, maize, rice, and wheat), which, combined, contribute more than half of all calories consumed by humans and cover more than 45% of global cropland (FAO, 2014). Our objectives were to: (1) examine soil GHG emissions and crop yield with NT soil management in various environmental and management conditions; (2) identify factors contributing to food security and climate change mitigation in support of NT as an effective CSA management practice.

**<u>Soil organic carbon</u>:** Soil organic carbon (SOC) is a primary indicator of soil health and plays a critical role in food production, greenhouse gas balance, and climate change mitigation and adaptation (Lorenz and Lal, 2016). The dynamic of agricultural SOC is regulated by the balance between carbon inputs (e.g., crop residues and organic fertilizers) and outputs (e.g., decomposition and erosion) under long-term constant environment and management conditions. However, this balance has been dramatically altered by climate change, which is expected to enhance SOC decomposition and weaken the capacity of soil to sequester carbon (Wiesmeier et al., 2016). Generally, agricultural soils contain considerably less SOC than soils under natural vegetation due to land conversion and cultivation (Hassink, 1997; Poeplau and Don, 2015), with the potential to sequester carbon from the atmosphere through proper management practices (Lal, 2018). Therefore, it is crucial to seek practical approaches to enhance agricultural SOC sequestration without compromising the provision of ecosystem services such as food, fiber, or other agricultural products.

The key to sequestering more carbon in soils lies in increasing carbon inputs and reducing carbon outputs. For example, conservation tillage practices, including no-tillage (NT) and reduced tillage (RT), are often recommended for SOC sequestration. Conservation tillage reduces soil disturbance and the soil organic matter decomposition rate (Salinas-Garcia et al., 1997) and promotes fungal and earthworm biomass (Lavelle et al., 1999; Briones and Schmidt, 2017), thereby improving SOC stabilization (Liang and Balser, 2012).

In recent decades, these management practices have been applied in major agricultural regions globally, and a large number of observations/measurements have been accumulated (e.g., Clark et al., 2017). However, their effects on SOC sequestration are variable and highly dependent on experiment designs and site-specific conditions such as climate and soil properties (Abdalla et al., 2016; Paustian et al., 2016). Some studies even suggested negative effects of CSA management practices on SOC (e.g., Liang et al., 2007). Also, most prior quantitative research focused on the effects of a single conservation tillage practice on SOC (e.g., Abdalla et al., 2016), very few studies estimated the combined effects of conservation tillage and other management practices like cover crops. Cover crops provide additional biomass inputs from above- and belowground (Blanco-Canqui et al., 2011), increase carbon and nitrogen inputs, and enhance the biodiversity of agroecosystems (Lal, 2004). Moreover, cover crops can promote soil aggregation and structure (Sainju et al., 2003), therefore indirectly reduce carbon loss from soil erosion (De Baets et al., 2011).

Some recent studies reported that a combination of cover crops and conservation tillage could significantly increase SOC compared to a single management practice (Blanco-Canqui et al., 2013; Ashworth et al., 2014; Duval et al., 2016; Higashi et al., 2014). For example, Sainju et al. (2006) suggested that soil carbon sequestration may increase 0.267 Mg C $ha^{-1}$ $year^{-1}$ under a combination of NT and cover crop practices, where the latter was a mixed culture of hairy vetch (*Vicia villosa*) and rye (*Secale cereale*); in contrast, a carbon loss of 0.967 Mg C $ha^{-1}$ $year^{-1}$ occurred when only NT was used. These findings highlight the importance of quantitatively evaluating the combined effects of multiple management practices on SOC sequestration under different climate and soil conditions.

This study aims to fill the abovementioned knowledge gap through a meta-analysis to simultaneously examine the effects of conservation tillage (i.e., NT and RT) on SOC sequestration (Figure 2.2). Our scientific objectives were to: (a) evaluate the effects of conservation tillage on SOC; (b) examine how environmental factors (e.g., soil properties and climate) and other agronomic practices (e.g., cover crops, nitrogen fertilization, residue management, irrigation, and crop rotation) influence SOC in conservation tillage environments.

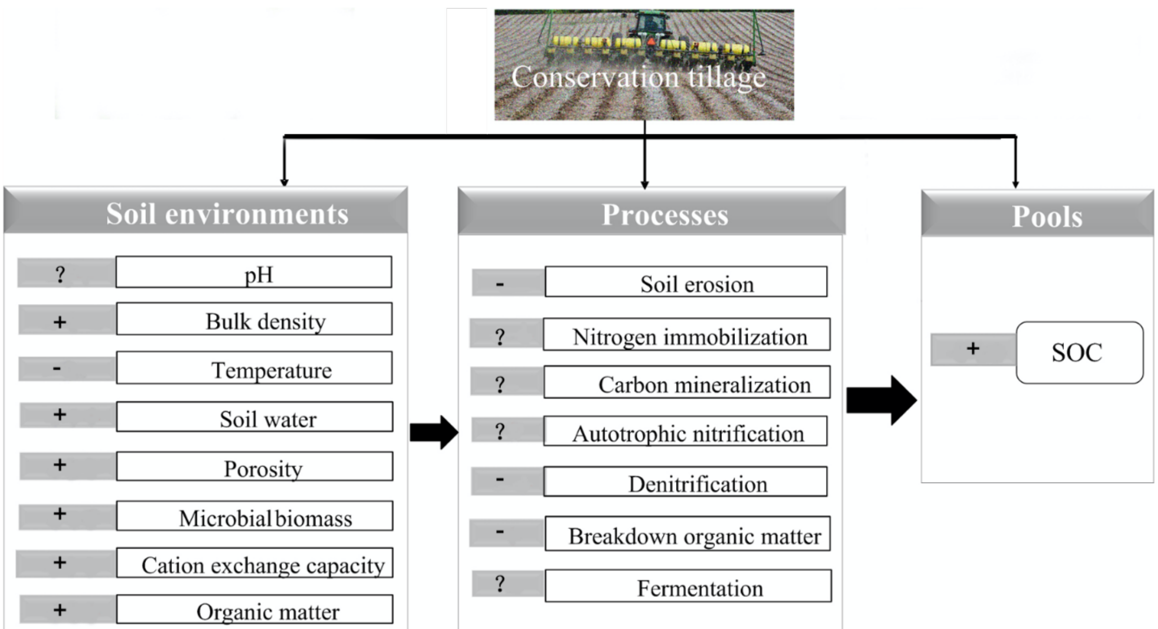


Figure 2.2 Relationship between conservation tillage and soil processes. "+" means positive feedback or promotion effect; "–" means negative feedback or inhibition function; and "?" means the effect is unclear

## 2.3 Materials and Methods

### 2.3.1 Data Compilation for Greenhouse Gas and Crop Yield Analysis

The data in this meta-analysis were collected from peer-reviewed publications reporting *in situ* soil GHG emissions and crop yield in both CT and NT soil management. A literature survey was performed using the Web of Science and Google Scholar (1900–

2017). Keywords used for the initial search included "tillage," "greenhouse gases," "$CO_2$," "$CH_4$,", and "$N_2O$." The literature survey focused on GHG emissions from four cereal crops (barley, maize, rice, and wheat). Three criteria were considered to minimize bias and ensure database quality when selecting studies. First, GHG emissions were measured *in situ* for the entire cropping season. Second, the CT versus NT comparison was done under otherwise similar agronomic management practices. Third, information regarding means, standard deviations (or standard errors), replications, and the magnitude of seasonal cumulative GHG emissions was either available or could be calculated.

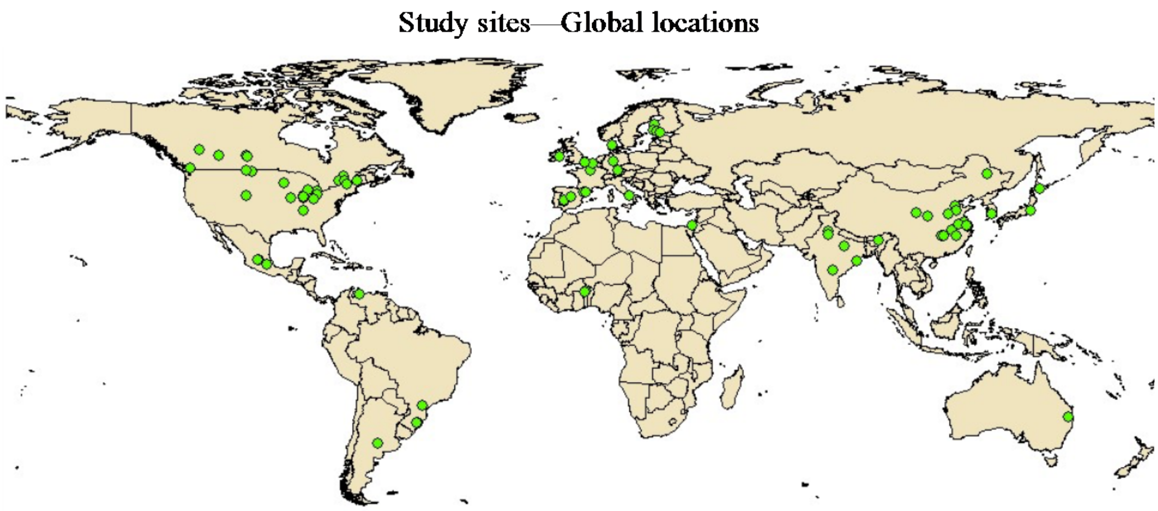


Figure 2.3 Global distribution of the study sites

Based on these criteria, 90 peer-reviewed articles with 139 comparisons for $CO_2$ emission, 65 for $CH_4$ emission, 56 for $CH_4$ uptake, 299 for $N_2O$ emission, and 181 for crop yield were collected for the meta-analysis (Data S2.1). These studies came from 20 different countries (Figure 2.3). The GHG and crop yield data were either derived from tables or extracted from figures using WebPlotDigitizer (Rohatgi, 2012). Other related information, including location (longitude and latitude), mean annual temperature (MAT), mean annual precipitation (MAP), land use, duration of the experiment, soil type, soil pH, crop residue management, and the rate and placement of N fertilizer inputs, was recorded. To disentangle the effects of other co-varying factors on GHG emissions and crop yield, we further analyzed data by considering two major categorical variables (i.e., environment and management), except when data availability constraints existed (Table 2.1). Climate regions were classified using global aridity values according to a generalized climate classification scheme (UNEP, 1997). The aridity index of each study site was extracted from the WorldClim database (Hijmans et al., 2005). Study sites with an aridity index > 0.65 were considered "humid," whereas study areas with a lower index (< 0.65) were grouped as "dry." Soil pH was classified into three categories following Havlin et al. (2013): acidic (pH < 6.6), neutral (6.6 ≤ pH ≤ 7.3), and alkaline (pH > 7.3). Soil texture was classified according to the USDA soil texture triangle. Clay, sandy clay, and silty clay classes were designated "fine textured;" silt, silt loam, silty clay loam, loam, sandy clay loam, and clay loam were considered "medium-textured;" and sand, loamy sand, and sandy loam were grouped as "coarse textured" (Daryanto et al., 2017a, b). Nitrogen (N) fertilization rates were grouped into four categories: control (no fertilizer applied), low (less than 100 kg N $ha^{-1}yr^{-1}$), medium (between 100 and 200 kg N $ha^{-1}yr^{-1}$), and high (greater than 200 kg N $ha^{-1}yr^{-1}$). Fertilizer N placement was grouped into "surface application" and "subsurface application." Methods such as injection, drilling, and side-dressing (depths of placement were clearly described in the literature) were considered to be subsurface N fertilizer applications. Crop residue management was classified as either "removed" or "retained." No-tillage management duration was determined according to NT establishment in each experiment. The NT treatment was considered "short" duration when imposed for less than five years, “medium” duration when present for 5 to 10 years, and “long” duration when exceeding ten years.

Table 2.1 Categories used in describing the environmental and management conditions

| Factors | Categories | | | |
|---|---|---|---|---|
| Environmental factors | | | | |
| Climate | Dry | Humid | | |
| Soil texture | Fine | Medium | Coarse | |
| Soil pH | Acid (<6.6) | Neutral (6.6-7.3) | Alkaline (>7.3) | |
| Management practices | | | | |
| Crop type | Rice | Wheat | Maize | Barley |
| Tillage duration | < 5 years | 5-10 years | $\geq$ 10 years | |
| N fertilizer | Control (0) | Low (<100 kg N $ha^{-1}yr^{-1}$) | Medium (100-200 kg N $ha^{-1}yr^{-1}$) | High ($\geq$ 200 kg N $ha^{-1}yr^{-1}$) |
| N placement | Surface (SUR) | Subsurface (SUB) | | |
| Crop residue management | Removed (RM) | Retained (RT) | | |

The variation in observed emission and yield was recorded and converted to the standard deviation (SD). The SD values were computed from standard error (SE) by the equation: SD = SE × $\sqrt{n}$ , where n is the number of replications. When SD and SE were missing, SD was estimated from the average coefficient of variation for the known data (Zhao et al., 2016).

### 2.3.2 Data Compilation for Soil Organic Carbon Analysis

We extracted data from 297 peer-reviewed articles published from 1990 to May 2017 (Data S2.2). Among all publications, 113 were conducted in the United States. All articles were identified from the Web of Science. The search keywords were “soil organic carbon” and “tillage.” All selected studies meet the following inclusion criteria: (a) SOC was measured in field experiments (to estimate the potential of biochar to increase soil carbon, we also included soil incubation and pot experiments with regard to biochar use); (b) observations were conducted on croplands excluding orchards and pastures; (c) ancillary information was provided, such as experiment duration, replication, and sampling depth; and (d) other agronomic management practices were included besides the three target management practices in this study. We considered conventional tillage as the control for NT and RT. Experiments that eliminated any tillage operation were grouped into the NT category, and experiments using tillage with lower frequency or shallower till-depth or less soil disturbance in comparison to the paired conventional tillage (e.g., moldboard plow and chisel plow) were grouped into the RT category.

SOC data were either derived from tables or extracted from figures using the GetData Graph Digitizer software v2.26 (http://getda ta-graph-digit izer.com/downl oad.php). Other related information from the selected studies were also recorded, including location (i.e., longitude and latitude), experiment duration, climate (mean annual air temperature and precipitation), soil properties (texture, depth, and pH), and other agronomic practices (crop residues, nitrogen fertilization, irrigation, and crop rotation). The study durations were grouped into three categories: short term (≤5 years), medium-term (6–20 years), and long term (>20 years). Climate was grouped according to the aridity index published by UNEP (1997) as either arid (≤0.65) or humid (>0.65). Study sites were grouped into cool (temperate and Mediterranean climates) and warm zones (semitropical and tropical climates; Shi et al., 2010). Soil texture was grouped as silt loam, sandy loam, clay and clay loam, loam, silty clay and silty clay loam, and loamy sand according to the USDA soil texture triangle. Soil depth was grouped as 0–10 cm, 10–20 cm, 20–50 cm, and 50–100 cm. Soil pH was grouped as acidic (<6.6), neutral (6.6–7.3), and alkaline (>7.3). Crop residue management was grouped as “residue returned” and “residue removed.” We only included those studies that used the same residue management in the control and treatment groups. Similarly, nitrogen fertilization was grouped into no addition, low (1–100 kg N/ha), medium (101–200), and high levels (>200). Irrigation management was grouped as irrigated or rainfed. Crop sequence was grouped as rotational or continuous crops (including crop-fallow systems). We also estimated the response of SOC in the whole-soil profiles (from the soil surface to 120 cm, with an interval of 10 cm) to tillage treatments.

The standard deviation (SD) of selected variables, an important input variable to the meta-analysis, was computed as $SD = SE \times \sqrt{n}$, where SE is the standard error, and n is the number of observational replications. If the results of a study were reported without SD or SE, SD was calculated based on the average coefficient of variation for the known data.

### 2.3.3 Data Analysis

A meta-analysis combines and compares results from pertinent independent studies by weighting these results according to their differences in precision. A random-effect meta-analysis was performed to explore environmental and management variables that might explain the response of GHG emissions and crop yield to NT, and the response of SOC to conservation tillage. In this meta-analysis, response ratios (R) comparing NT and CT, for GHG emissions and crop yield, were calculated as follows:

$$R = \left(\frac{X_t}{X_c}\right) \tag{2.1}$$

where X is the variate ($CO_2$, $CH_4$, $N_2O$, GWP, crop yield, or SOC) mean for either the treatment (NT/RT) or the control (CT) treatment. The natural logarithm of R (*lnR*), the effect size, was calculated for each treatment in every trial/experiment (Hedges et al., 1999; Deng et al., 2017). The variance ($\upsilon$) of *lnR* was computed as:

$$\upsilon = \frac{SD_t^2}{n_{NT}X_t^2} + \frac{SD_c^2}{n_{CT}X_c^2} \tag{2.2}$$

where SD and n are standard deviation and sample sizes, respectively, either in CT or NT. The weight of each effect size was:

$$\omega = \frac{1}{\upsilon} \tag{2.3}$$

The mean effect sizes were estimated as:

$$\overline{lnR} = \frac{\sum(lnR_i \times \omega_i)}{\sum \omega_i} \tag{2.4}$$

where $lnR_i$ and $\omega_i$ were the effect size and weight from the ith comparison, respectively. The 95% confidence interval (CI) of $\overline{lnR}$ was computed as:

$$95\%CI = \overline{lnR} \pm 1.96SE_{\overline{lnR}} \tag{2.5}$$

where $SE_{\overline{lnR}}$ is the standard error of $\overline{lnR}$ and was computed as:

$$SE_{\overline{lnR}} = \sqrt{1/\sum \omega_i} \tag{2.6}$$

SAS software was used to analyze the data by applying the macros for the meta-analysis procedure (Lipsey and Wilson, 2001). The effect size means were significantly different if their 95% CI did not overlap with zero. The percent change in selected variables was computed using the equation:

$$\left(e^{\overline{lnR}} - 1\right) \times 100\% \tag{2.7}$$

Global warming potential was calculated when fluxes for all three GHG species (i.e., $CH_4$, $CO_2$, and $N_2O$) were reported in a single study. The units of soil $CH_4$ and $N_2O$ fluxes were converted into $CO_2$-equivalent units before GWP calculation. We used the IPCC factors (IPCC, 2013) to calculate GWP in $CO_2$-equivalents $ha^{-1}$ $yr^{-1}$ over a 100-year time horizon:

$$\mathrm{GWP} = \mathrm{CO_2} \times 1 + \mathrm{CH_4} \times 34 + \mathrm{N_2O} \times 298 \tag{2.8}$$

Each categorical environment and management variable was treated as a moderator in analyzing the whole dataset. The Chi-square test was then used to calculate the between-group heterogeneity for a given variable across all the data to further analyze the NT effect for different sub-categories. Publication bias was tested by the funnel plot method and assessed using Kendell's rank correlation (Begg and Mazumdar, 1994). If the mean effect exhibited a significant difference from zero (i.e., indicating publication bias), Rosenthal's fail-safe or file drawer number was calculated (METAFOR package in R) to estimate if our conclusion was likely affected by nonpublished studies (Rosenberg, 2005). The meta-analysis can be considered robust if the fail-safe number is larger than 5 × k + 10 (where k is the number of observed studies; Rothstein et al., 2005).

## 2.4 Results

### 2.4.1 Greenhouse Gas Emissions and Crop Yield

#### 2.4.1.1 Overall Effects of NT on GHG Emissions and Crop Yield

On average, the $CO_2$ emission rate was not significantly different between NT and CT (CI overlapped with zero; Figure 2.4). In contrast, NT significantly increased $N_2O$ emission by 10.4% with a mean weighted *lnR* of 0.10 [CI = (0.02, 0.17)] (Figure 2.4). For locations exhibiting net $CH_4$ uptake, we found no difference between NT and CT. Whereas for sites with net $CH_4$ emission, NT reduced $CH_4$ emission by 15.5%, with an *lnR* of -0.17 [CI = (-0.30, -0.03)] (Figure 2.4). Therefore, a reduction in $CH_4$ emission was the major contribution of NT management to GHG mitigation. Crop yields were similar between NT and CT (CI overlapped with zero, Figure 2.4), suggesting that yield loss should not be a deterrent to NT adoption as a CSA practice. Publication bias for $CH_4$ uptake analysis was suggested by Rosenthal's fail-safe number method, but it was not found for the other variables (Table S2.1).

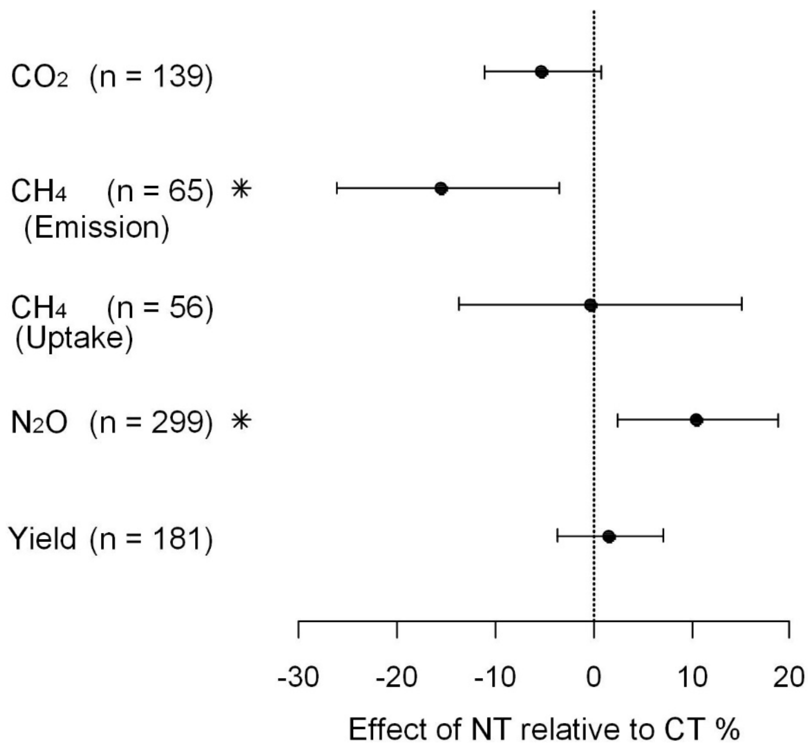


Figure 2.4 Overall changes in soil greenhouse gas (GHG) emissions and crop yields between NT and CT. Numerals indicate the number of observations. * represents $p < 0.05$

Our analysis revealed several environment and management variables affecting GHG emissions and crop yield in NT (versus CT) management. The response of soil $CO_2$ emission to NT varied significantly with crop species, climate regime, NT duration, soil pH, and crop residue management. No-tillage-induced changes in $CH_4$ flux were significantly influenced by crop species, soil pH, and NT duration. No-tillage-induced changes in $N_2O$ flux were significantly impacted by N fertilizer placement. Differences in crop yield due to NT were significantly related to crop species, climate, and both N fertilizer rate and placement (Table 2.2).

Table 2.2 Summary of the Chi-square test for the variables controlling the comparative effect of tillage (no-tillage vs. conventional tillage) on GHG emissions and crop yield

| | $CO_2$ | | $CH_4$ | | | $N_2O$ | | GHG | | Yield | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Variables | df | $\chi^2$ | df | $\chi_1^2$ | $\chi_2^2$ | df | $\chi^2$ | df | $\chi^2$ | df | $\chi^2$ |
| Crop types | 3 | 18.24*** | 2 | 9.56** | 7.76* | 3 | 1.14 | 3 | 13.77** | 3 | 42.96*** |
| Climate | 1 | 5.42** | 1 | 1.12 | 4.9* | 1 | 0.22 | 1 | 0.2 | 1 | 10.95*** |
| Duration | 1 | 13.96*** | 1 | 18.11*** | 5.25 | 2 | 0.23 | 1 | 2.69 | 2 | 2.69 |
| Texture | 2 | 1.07 | 2 | 0.74 | 1.71 | 2 | 5.92 | 2 | 2.57 | 2 | 1.85 |
| pH | 2 | 11.71** | 2 | 9.57** | 18.4*** | 2 | 2.42 | 2 | 7.6* | 2 | 5.91 |
| N rate | 3 | 0.53 | 3 | 3.88 | 6.9 | 3 | 2.56 | 3 | 2.21 | 3 | 10.08* |
| N placement | 1 | 0.6 | 1 | 0.13 | 0.56 | 1 | 10.44** | 1 | 0.21 | 1 | 5.09* |
| Residue | 1 | 4.04* | 1 | 0.63 | 2.46 | 1 | 0.15 | 1 | 0.8 | 1 | 0.16 |

df represents degrees of freedom. $\chi_1^2$ represents $CH_4$ emissions, $\chi_2^2$ represents $CH_4$ uptakes.

Statistical significance: *P< 0.05; **P< 0.01; ***P< 0.001.

#### 2.4.1.2 Effects of Environment on NT vs. CT Comparison

##### 2.4.1.2.1 Climate

Compared to CT, NT significantly decreased $CO_2$ emission (-9.9%) in dry climates, but not in humid climates (Figure 2.5a). However, significant reductions in soil $CH_4$ emission (-18.9%) with NT occurred in humid climates (Figure 2.5b). Similarly, NT increased soil CH4 uptake (47%) in humid climates (Figure S2.1a). Although climate did not significantly affect the difference in $N_2O$ emission between the tillage practices (Table 2.2), $N_2O$ emission in NT was greater (12.3%) than in CT for humid climates (Figure 2.5c). The effect of NT on crop yield in different climates was not consistent. In arid climates, NT soil management caused 10.2% greater crop yield, but in humid climates, NT decreased yield by 7.5%, compared to CT (Figure 2.5d).

##### 2.4.1.2.2 Soil Texture and pH

Generally, soil texture had no statistically significant effect on the difference between NT and CT in terms of soil GHG emissions or crop yield (Table 2.2). In fine-textured soils, NT resulted in significantly higher (32.2%) $N_2O$ emission (Figure 2.4c). With NT, $CO_2$ emission was significantly lower, by 15.3% and 18.4%, in acidic and neutral soils, respectively (Figure 2.4a). No-tillage also significantly reduced $CH_4$ emission by 25.2% in acidic soils (Figure 2.4b). We found a significant decrease (31.2%) in $CH_4$ uptake in neutral soils with NT (Figure S2.1a). Differences in $N_2O$ emissions between NT and CT were not significantly influenced by soil pH, although there was a nearly 15% greater emission from acidic soils in NT (Figure 2.5c, Table 2.2). For crop yield, our results suggested that NT improved yield (13.1%) in alkaline soils (Figure 2.5d).

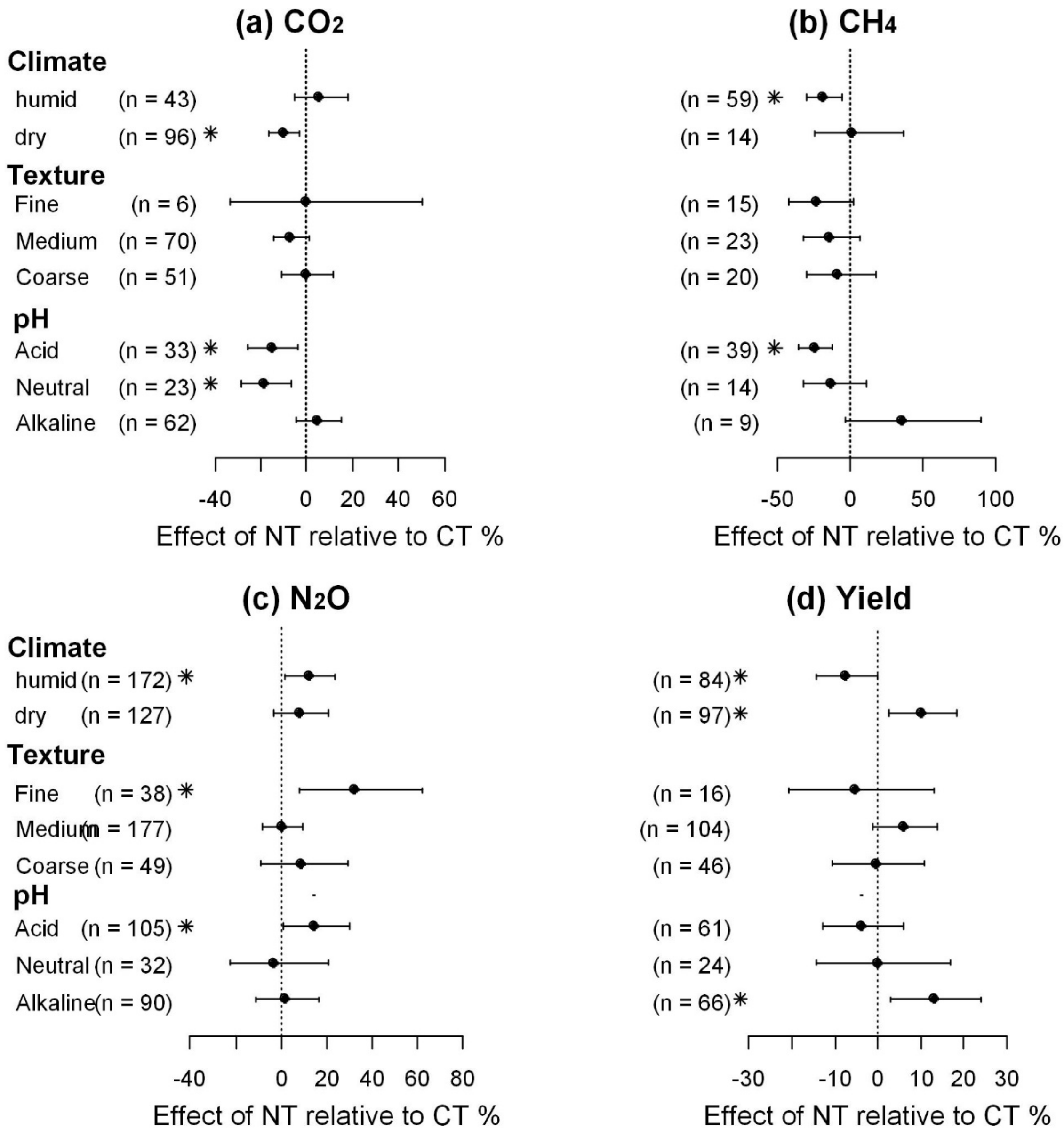


Figure 2.5 The effect of NT on soil greenhouse gas (GHG) emissions and crop yields differed with environmental factors (a) $CO_2$ (b) $CH_4$ (c) $N_2O$ (d) Yield. Numerals indicate the number of observations. * represents $p < 0.05$

#### 2.4.1.3 Interactive Controls of Management Practices with CT and NT

##### 2.4.1.3.1 Nitrogen Fertilization

Nitrogen fertilizer application rate did not significantly affect the difference in soil GHG emissions between NT and CT (Table 2.2). Compared to CT, NT significantly reduced $CH_4$ emission (24.7%) at the medium N fertilizer rate (Figure 2.6b), and it resulted in significantly higher $N_2O$ emission (16.7%) at the high fertilizer rate (Figure 2.6c). However, N fertilizer rate played a significant role in crop yield differences between NT and CT (Table 2.2). Specifically, NT enhanced crop yield (25.2%) at the low fertilizer N rate (Figure 2.6d). Additionally, though the NT versus CT response patterns in $N_2O$ emission and crop yield due to N fertilizer placement were similar (Figure 2.6c, d), there was a significant difference between the two fertilization sub-groups (Table 2.2). Compared to CT, surface N fertilizer placement in NT exhibited 18.6% higher $N_2O$ emission (Figure 2.6c). Changes in $N_2O$ emission due to the tillage practices were not significant with subsurface N placement. Fertilizer N placement had no effect on the differences in $CO_2$ emission or $CH_4$ flux between the tillage practices (Table 2.2).

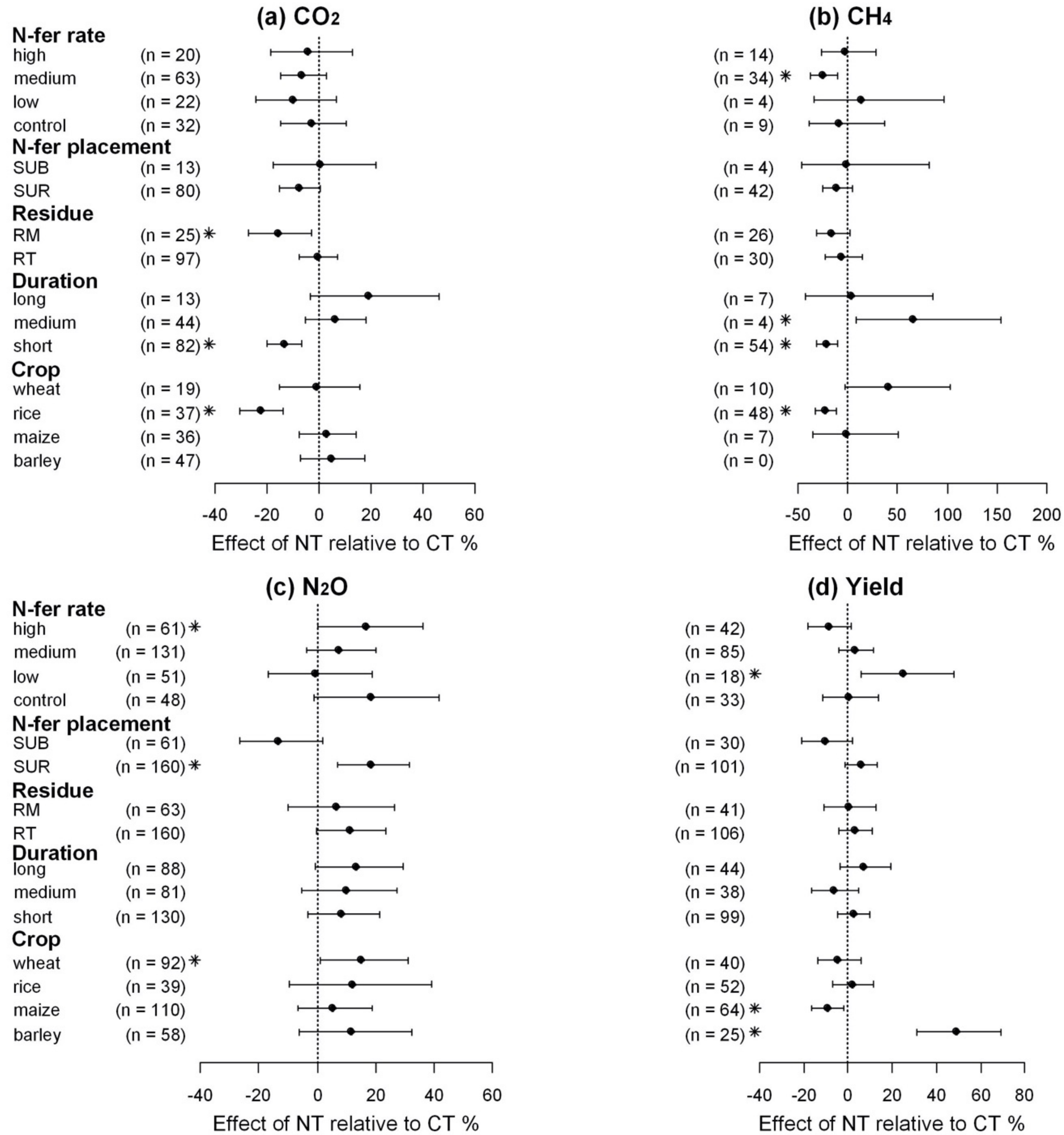


Figure 2.6 The effect of NT on soil greenhouse gas (GHG) emissions and crop yields differed with management factors. (a) $CO_2$ (b) $CH_4$ (c) $N_2O$ (d) Yield. Numerals indicate the number of observations. * represents $p < 0.05$

#### 2.4.1.3.2 Residue Management and Duration of No-Tillage

Crop residue management only significantly affected $CO_2$ emission (Table 2.2). With crop residue removal, $CO_2$ emission was significantly lower (-15.8%) in NT (Figure

2.6a). However, with residue retention, there was no significant difference in $CO_2$ emission between the tillage practices. Changes in $CH_4$ and $N_2O$ fluxes and crop yield due to NT were not influenced by residue management. The duration of NT had inconsistent effects on $CO_2$ and $CH_4$ emissions. Emissions of $CO_2$ and $CH_4$ were reduced by 13.3% and 21.4%, respectively, with short NT duration (Figure 2.6a, b), but there was no difference between NT and CT for studies where NT duration was longer than five years. The duration of NT had no impact on the differences in $CH_4$ uptake, $N_2O$ emission, or crop yield.

#### 2.4.1.3.3 Crop Species

The crop species being grown played a significant role in the differences in GHG emissions and crop yield due to tillage (Table 2.2). The difference in $CO_2$ emission between NT and CT was largest with rice, where NT soils emitted 22.5% less $CO_2$ than CT soils (Figure 2.6a). No-tillage also reduced $CH_4$ emission in rice production systems by 22.4% (Figure 2.6b) but increased $CH_4$ uptake in wheat by 31.1% (Figure S2.1b). There were no significant differences in $CO_2$ emission and $CH_4$ flux between tillage practices in barley or maize production systems. Changes in $N_2O$ emission between NT and CT were only significantly different in wheat production, where a 15.2% increase in emission under NT was noted (Figure 2.6c). Crop yield differences between NT and CT were significant in barley and maize production systems, with barley yield being 49% higher and maize yield 9.3% lower under NT (Figure 2.6d).

### 2.4.1.4 Effects of No-Tillage on Global Warming Potential

For those studies that measured fluxes of all three GHGs, NT exhibited no difference in GWP compared to CT (Figure 2.7). Further examination of the relevant environment and management variables showed that soil pH and crop species significantly affected the difference in GWP between tillage systems (Table 2.2). No-tillage decreased GWP by 31.2% in acidic soils and by 24.8% in rice fields (Figure 2.7). This pattern generally matched the effect of NT on $CO_2$ flux (Figure 2.5 and 2.6). However, with the realization that fluxes of all three GHGs and crop yield were reported in few studies comparing NT and CT, these results were likely affected by publication bias (Table S2.1), and therefore should be interpreted cautiously.

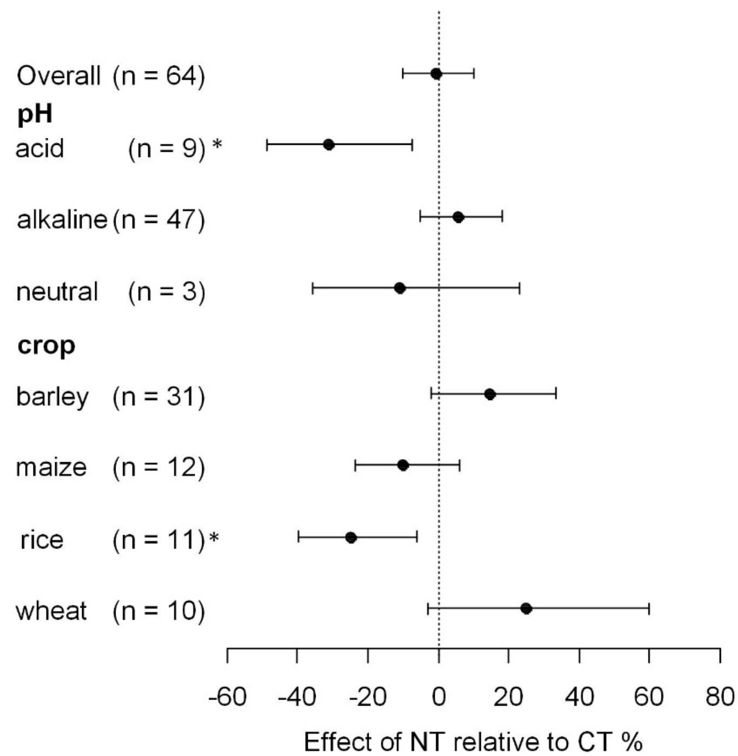


Figure 2.7 The effect of NT on the global warming potential (GWP) of greenhouse gas (GHG) emissions. Numerals indicate the number of observations. * represents $p < 0.05$

### 2.4.2 Soil Organic Carbon

#### 2.4.2.1 Overall SOC Responses to Conservation Tillage

Overall, conservation tillage enhanced SOC storage by 5% (Figure 2.8). When investigating different types of conservation tillage, NT and RT had similar effects on SOC (approximately 8% increase). All results were statistically significant (Figure 2.8). Across the whole dataset we compiled, the SOC varied widely in each tillage treatment (Figure S2.2). We calculated the distribution of the data points (the ratio of SOC of each treatment to that of the corresponding control, i.e., NT/RT vs. conventional tillage; Figure S2.2). Most of the studies used in this meta-analysis reported positive responses of SOC to NT and RT (60% and 65%, respectively). The SOC change rates were $0.38 \pm 0.71$ Mg ha$^{-1}$ year$^{-1}$ ($n = 56$) and $-0.29 \pm 0.79$ Mg ha$^{-1}$ year$^{-1}$ ($n = 30$) in NT and RT systems, respectively (Figure S2.3).

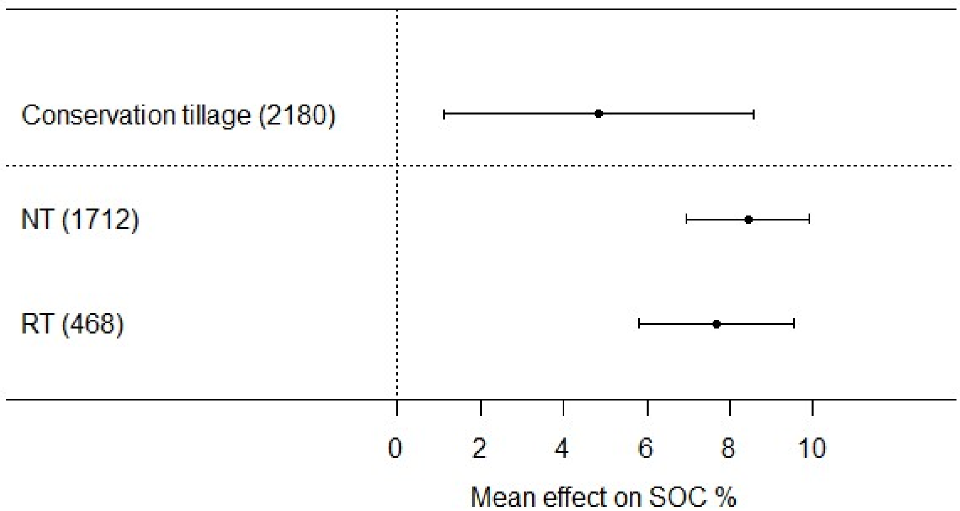


Figure 2.8 Overall changes in soil organic carbon (SOC) between conservation tillage (no-tillage (NT) and reduced tillage (RT)) and conventional tillage. Error bars represent 95% confidence intervals

#### 2.4.2.2 Effects of Conservation Tillage in Different Climate and Soil

*Climate:* Overall, conservation tillage sequestered more SOC in arid areas than in humid areas (Figure 2.9a). The NT-induced SOC increase was slightly higher in arid areas than that in humid areas (9% and 8%, respectively). In comparison, the RT-induced SOC increment in arid areas was two times greater than that in humid areas. Our further analysis suggested that conservation tillage significantly increased SOC in both cool and warm climate zones with diverse responses (Figure 2.9b). In warm areas, NT increased SOC by 15% compared to 8% in cool areas. Reduced tillage increased SOC by 7% and 6% in warm and cool areas, respectively.

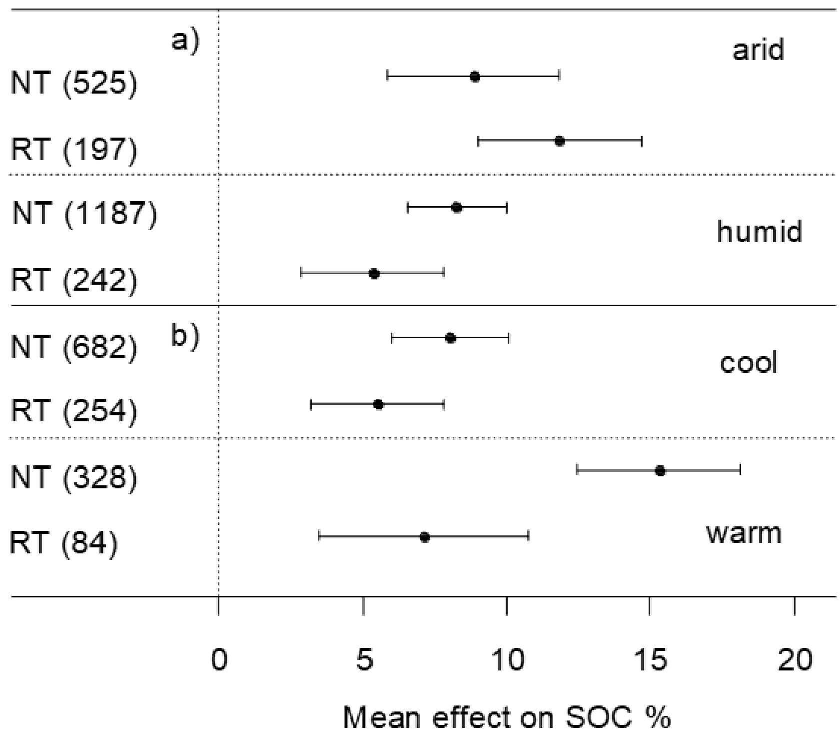


Figure 2.9 The effect of NT (no-tillage) and RT (reduced tillage) on soil organic carbon (SOC) differed with climate zones (the climate zones were divided by a) aridity index; b) mean annual air temperature). Numbers in parentheses represent the number of observations. Error bars represent 95% confidence intervals

*Soil texture:* The effects of conservation tillage on SOC were strongly influenced by soil texture (Figure 2.10a). No-till increased SOC by 16% in silty clay and silty clay loam soils, compared to 12% in sandy loam soils and 7% in loamy sand soils. Reduced tillage increased SOC by 21%, 7%, and 15% in silty clay and silty clay loam soils, loam soils, and loamy sand soils, respectively. Overall, NT and RT increased SOC more in fine-textured soils than in coarse-textured soils.

*Soil depth:* The positive effects of conservation tillage on SOC decreased with soil depth (Figure 2.10b). Both NT and RT could significantly increase SOC most at 0–10 cm depth (22% and 17%, respectively). Although reduced SOC was observed in the 10–20 cm and 20–50 cm soil layers (−4% and −10%, respectively), NT could still enhance SOC sequestration in the entire soil profile up to 120 cm (Table S2.5). In comparison, RT could increase SOC in the 0–70 cm soil profile (Table S2.5), although decreased soil carbon (not statistically significant) was observed in the 10–50 cm soil layer (Figure 2.10b).

*Soil pH:* The management-induced SOC uptake was generally higher in alkaline soils than in acid soils (Figure 2.10c). No-till increased SOC by 6% in acid soils and 13% in alkaline soils. The SOC increased by RT was greater in alkaline soils (9%) than in acid soils (6%), but RT had no significant influence on SOC in neutral soils.

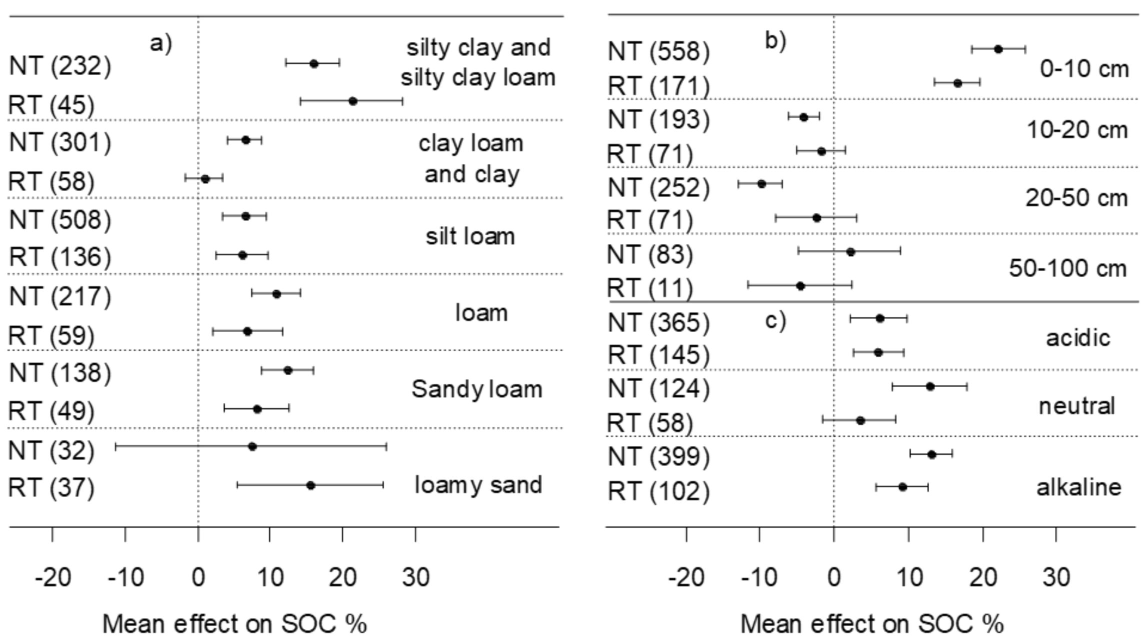


Figure 2.10 The effect of NT (no-tillage) and RT (reduced tillage) on soil organic carbon (SOC) differed with soil texture (a), soil depth (b), and soil pH (c). Numbers in paratheses represent the number of observations. Error bars represent 95% confidence intervals. The average depth of each categorial group was presented in supplementary files (Tables S2.8-S2.11)

#### 2.4.2.3 Combined Effects of Experiment Duration and Other Agronomic Practices

Conservation tillage practices are generally applied together with other agronomic practices such as residue return, nitrogen fertilizer use, and irrigation. These agronomic practices may interact with conservation tillage practices with positive or negative effects on the capacity of soils to sequester carbon. In this study, we considered experiment duration and five other agronomic practices, including residue return, nitrogen fertilization, irrigation, crop sequence, and cover crops, to quantify these effects.

*Duration:* Our results demonstrated that NT significantly increased SOC by 13% in the long-term experiments, followed by medium-term (7%) and short-term experiments (6%; Figure 2.11a). Reduced tillage increased SOC by 12% in long-term studies, followed by medium-term (9%) and short-term experiments (3%). The average durations differed in each group (Table S2.6), which may influence the effect of CSA management practices on SOC. When excluding short and medium experiment durations (≤20 years) and shallow sampling (<20 cm), RT significantly increased SOC by 14%, while NT had no significant effect on SOC (Figure S2.4).

*Residues:* When crop residues were returned, conservation tillage significantly increased SOC: 9% for NT and 5% for RT (Figure 2.11b). However, if crop residues were removed, RT had a significant effect on SOC, although there was a significant increase in SOC under NT (5%).

*Nitrogen fertilizer:* Our results suggested that nitrogen fertilizer use could alter the magnitude of soil carbon uptake induced by conservation tillage practices. No-till tended to sequester more soil carbon when nitrogen fertilizer input was relatively lower (11%, 8%, and 6% for low-level, medium-level, and high-level nitrogen fertilization, respectively).

While RT increased SOC by 13% at the medium-level nitrogen fertilizer rate, approximately two times larger than those at the low-level and high-level nitrogen fertilizer use (Figure 2.11d).

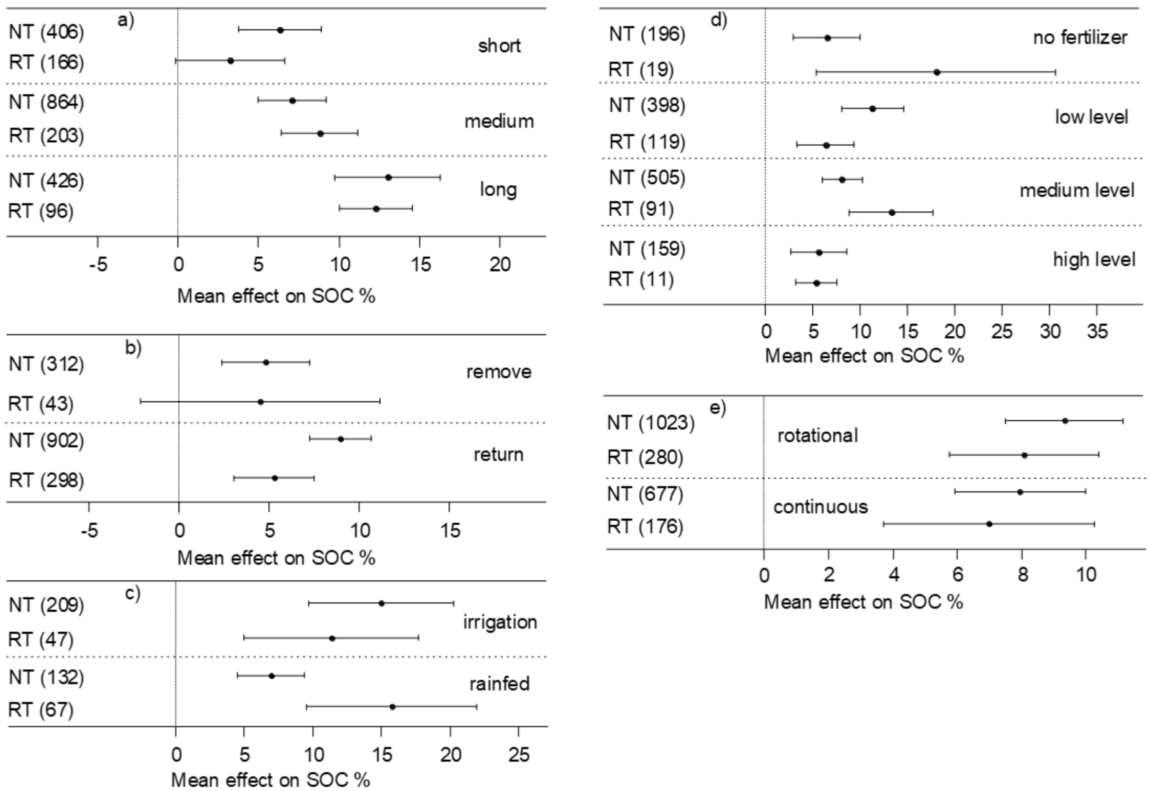


Figure 2.11 The effect of NT (no-tillage) and RT (reduced tillage) on soil organic carbon (SOC) differed with experiment duration (a), residue management (b), water management (c), nitrogen fertilizer use (d), and crop sequence (e). Numbers in paratheses represent the number of observations. Error bars represent 95% confidence intervals

*Irrigation:* When investigating the irrigation effects, our results suggested that NT increased SOC by 15% in irrigated croplands, twice as much soil carbon as that in rainfed croplands. In contrast, the RT-induced SOC increase was 16% under the rainfed condition, 5% higher than that in irrigated croplands (Figure 2.11c).

*Crop rotation:* Conservation tillage significantly promoted SOC in both rotational and continuous cropping systems (Figure 2.11e). No-tillage and RT induced SOC increases showed no obvious differences in the rotational and continuous cropping systems (9% and 8% vs. 8% and 7%).

*Cover crops:* Our results demonstrated that combining conservation tillage and cover crops might significantly enhance SOC sequestration. In warm regions, SOC increased by 13% with the combination of conservation tillage and cover crops (Figure 2.12). In loamy sand and sandy clay loam soils, associated SOC uptakes increased to 31% and 21%, respectively. A similar effect was also observed in medium-term experiments. However, in clay soils, the combination of cover crops and conservation tillage significantly decreased SOC by 19%.

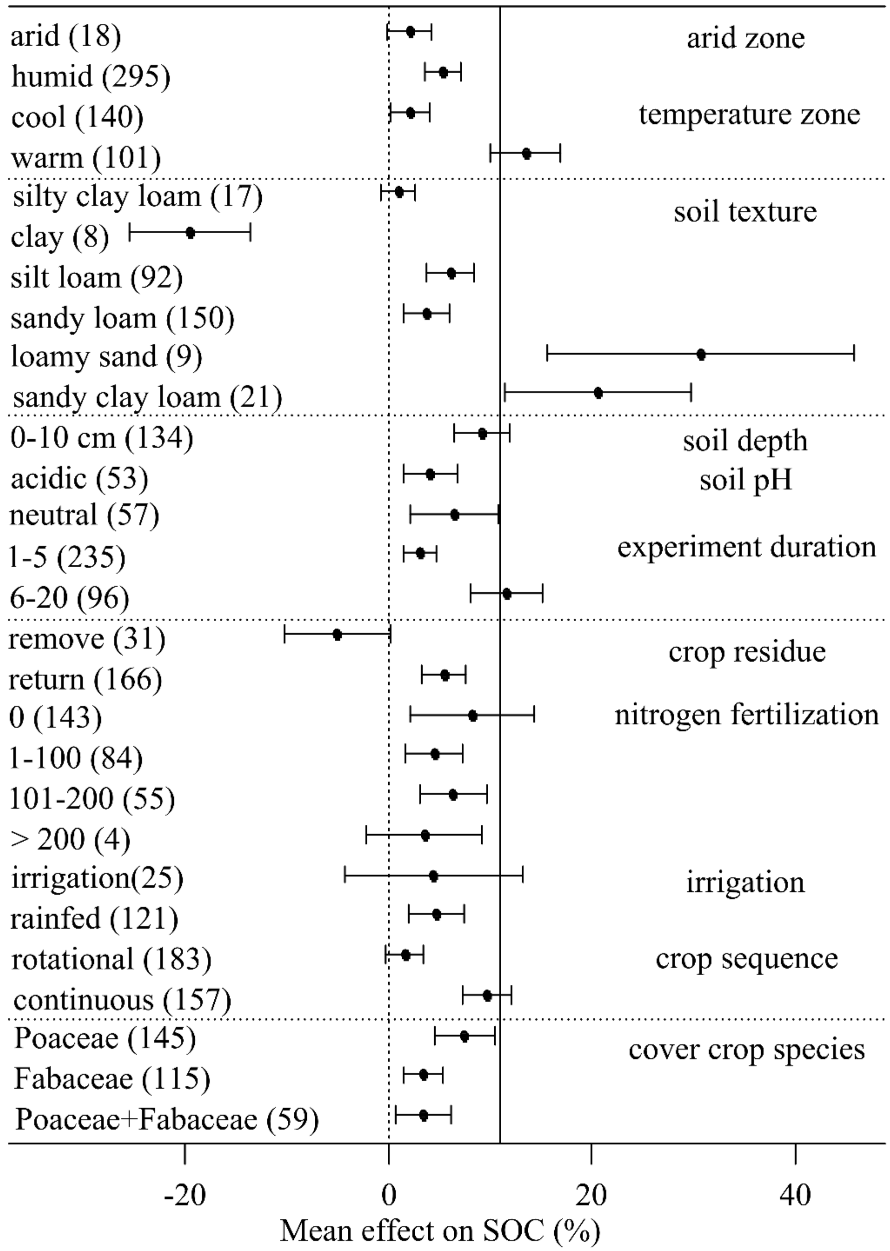


Figure 2.12 The combined effect of conservation tillage and cover crops on SOC (soil organic carbon) for different subcategories. Numbers in paratheses represent the number of observations. Error bars represent 95% confidence intervals. The vertical solid line represents 11%, which is the theoretical sum of the effect sizes of conservation tillage and cover crops (Bai et al., 2019)

## 2.5 Discussion

### 2.5.1 Responses of GHG Emissions to NT

Our meta-analysis found no significant effect of NT on $CO_2$ emission. This contrasts with the results of an earlier meta-analysis documenting a significant decrease in $CO_2$ emission (-21%) with NT (Abdalla et al., 2016). This discrepancy might be related to data source differences. In Abdalla et al. (2016), data were collected from experiments in which $CO_2$ emission was only measured for a period immediately after tillage - not for the entire growing season. This shortened measurement period may have amplified the impact of tillage on $CO_2$ emission, as it possibly captures the release of $CO_2$ previously trapped in soil pores (Oorts et al., 2007). The immediate stimulation of tillage on $CO_2$ production was likely due to the breakdown of aggregates and exposure of otherwise protected SOM (Fiedler et al., 2016). Thus, short-duration studies might not be sufficient to capture the magnitude of $CO_2$ emission associated with season-long decay of surface crop residues in NT (Oorts et al., 2007). The response of GHG fluxes to NT varies considerably with GHG flux measurement timing (Regina and Alakukku, 2010). Although NT management has often been touted to reduce $CO_2$ emission (Kessavalou et al., 1998), greater $CO_2$ emission in NT has also been reported. This is likely due to the decomposition of crop residues accumulated in long-term NT (Oorts et al., 2007) or to enhanced soil respiration by a more abundant soil microbial population (Plaza-Bonilla et al., 2014b).

No-tillage decreased soil $CH_4$ emission by 15.5% and had no effect on soil $CH_4$ uptake, which is consistent with another meta-analysis reported for Chinese rice paddies (Zhao et al., 2016). Generally, rice paddies act as atmospheric $CH_4$ sources, while upland soils are either $CH_4$ sinks or sources, depending on the balance between soil methanogenic and methanotrophic activities (Topp and Pattey, 1997). Soil properties such as SOC, temperature, and bulk density play a leading role in controlling the activity of methanogens and methanotrophs, affecting the direction of $CH_4$ flux (Mitra et al., 2002). On the one hand, NT results in higher surface SOC, soil water content and bulk density (Ahmad et al., 2009; Zhang et al., 2015), thus increasing the potential for $CH_4$ production due to greater availability of organic substrates and formation of anaerobic microsites. On the other hand, NT increases soil macroporosity and soil pore continuity (Ball et al., 1999), thus improving gas diffusivity and increasing $CH_4$ oxidation.

Our meta-analysis found that NT significantly increased soil $N_2O$ emission by 10.4%. Higher $N_2O$ emission in NT is usually ascribed to enhanced soil microbial activities, especially denitrification, due to increased soil moisture and decreased soil aeration (Venterea et al., 2005; Almaraz et al., 2009b; Ma et al., 2013). Our results were different from some other studies. For example, Gregorich et al. (2008) observed higher N2O emission from CT soils, and these authors suggested that nitrification ($NO_3^-$ formation) was controlling $N_2O$ emission from soils in CT due to greater soil aeration and lower soil water content. The contradictory findings might result from different microbial activity in responses to site-specific conditions. The microbial community may vary from site to site and interact with NT management, leading to different responses of $N_2O$ emission.

### 2.5.2 Factors in Regulating GHG Emissions

#### 2.5.2.1 Environmental Factors

Climate greatly influenced the differences in GHG emissions between tillage practices (Table 2.2). For instance, NT strongly reduced $CO_2$ emission in dry climates. Similar trends in $CO_2$ emission in NT have been reported (Abdalla et al., 2016). The larger difference between NT and CT in dry climates can be attributed to the differences in soil temperature (Lu et al., 2016) and soil water availability (Álvaro-Fuentes et al., 2008). No-tillage normally causes greater soil water content than does CT (Abdalla et al., 2013). The resulting difference in soil moisture between tillage practices tends to be large at dry sites (Feiziene et al., 2012), and so does the difference in soil temperature (Lu et al., 2016). In terms of the tillage effects on $CH_4$ emission, although there was a great reduction with NT in humid climates, the climate regime had no significant influence (Table 2.2). A 90% decrease in the NT soil $CH_4$ emission was observed by Sapkota et al. (2015) in a semi-arid rice field, which was largely attributed to a different water management strategy that caused a shorter flooding period in the NT plots. Continuous flooding can be a major factor controlling $CH_4$ production because as soil redox potential falls below −150 mV, methanogenesis is favored (Masscheleyn et al., 1993). Considering that NT soils maintain an improved moisture regime, irrigation schedules with shortened flooding periods can be expected to reduce rice field $CH_4$ emission, regardless of the climate regime. In regard to $N_2O$, there was a significantly higher emission with NT, relative to CT, in humid climates. Humid climates, which exhibit higher precipitation frequency, together with greater NT soil moisture, promote denitrification driven $N_2O$ emission. Total $N_2O$ emission depends on how long favorable conditions persist (Hunt et al., 2016). This is supported by Almaraz et al. (2009b), who found that precipitation is the major driver of $N_2O$ emission and that difference between NT and CT in $N_2O$ emission is significantly larger in a wet year than a dry year.

The significant increase in $N_2O$ emission in fine-textured NT soils (Figure 2.4c) is noteworthy and is consistent with previous studies (Rochette, 2008). Soil texture may modify the effects of NT on $N_2O$ emission via differences in soil water content (Abdalla et al., 2013). Rochette (2008) reported increased $N_2O$ emission from poorly-drained fine-textured soils under NT located in regions with humid climates. Implementing NT in fine-textured soils in humid areas increased $N_2O$ emission by approximately 38% compared to CT (Table S2.2). This indicates that climate-soil interactions should be considered before adopting NT as a CSA practice in a certain region.

The impact of NT on GHG emissions was sensitive to soil acidity. In acidic soils, $CO_2$ and $CH_4$ emissions were reduced, but $N_2O$ emission increased with NT management. Soil pH can affect GHG production in soils, and pH can also be affected by different tillage regimes. Microbial activity, the major source of soil $CO_2$ emission and often globally expressed as respiration is sensitive to soil pH. Increased basal respiration with increased pH has been widely reported (Lundström et al., 2003). As NT management is known to result in reduced topsoil pH (Dick, 1983), decreased dissolved organic carbon and $CO_2$ emission is expected. The optimum soil pH for $CH_4$ production is near neutrality. Considering methanogenic bacteria are acid-sensitive, a small decrease in soil pH can substantially reduce $CH_4$ production, whereas a slight increase in soil pH can produce the opposite response (Wang et al., 1993). Higher $N_2O$ emission from acidic soils can be ascribed to the greater sensitivity of $N_2O$ reductase to low pH than that of the other denitrification reductases (Thomsen et al., 1994). This can result in a higher ratio of $N_2O$ to $N_2$ as pH declines, and therefore greater $N_2O$ loss from low pH soils (Baggs et al., 2010).

#### 2.5.2.2 Management Factors

Differences in $CO_2$ emission between NT and CT did not differ with N fertilizer rate or placement, which agreed with Plaza-Bonilla et al. (2014b) and Snyder et al. (2009).

Similar results were reported by Abdalla et al. (2016) and may be attributed to the overriding impact of N fertilization in enhancing productivity and carbon inputs to both NT and CT soils. Similarly, N fertilization did not significantly alter the differences in $CH_4$ emission between the tillage practices. However, N fertilization is considered the main stimulus to increased agroecosystem $N_2O$ emission (Grace et al., 2011). A sharp rise in $N_2O$ emission within days of fertilization is commonly observed in both NT and CT (Halvorson et al., 2008; Sapkota et al., 2015). Compared with CT, soil environmental and physical conditions in NT are expected to be conducive to greater denitrifying activity and a greater likelihood of $N_2O$ emission following surface application of N fertilizer (Venterea et al., 2005). Soil organic matter and microbial population are usually more uniformly distributed with depth in CT. The vertical distribution of potential denitrifying activity varies with tillage, with higher facultative anaerobe populations and potential denitrification rates in the topsoil of NT compared to CT (Linn and Doran, 1984a; Groffman, 1985).

The surface placement of N fertilizer likely provides adequate substrate to the more abundant population of denitrifiers in the NT soil surface. This, together with a wetter and denser soil environment, enhances denitrifying $N_2O$ emission. With subsurface N fertilizer application, less $N_2O$ emission under NT could result from lower denitrifier populations and/or available C concentration at the greater depth (Drury et al., 2006), relative to CT soils. The greater water-filled pore space observed in NT may also increase the probability of reduction of $N_2O$ to $N_2$ during upward diffusion (Linn and Doran, 1984b), further reducing $N_2O$ emission with subsurface N fertilizer placement.

Tillage is often associated with residue management. There were indirect effects of residue management on differences in GHG emissions between the tillage practices. With residue removal, the reduction in $CO_2$ emission was greater in NT than CT. This was expected considering that NT has little effect on the SOM turnover rate, while CT accelerates SOM turnover via thorough surface soil disturbance. However, large uncertainties in the responses of GHG emissions exist when considering the opposite operation, residue retention, as both residue quantity and quality are important to soil physical and chemical properties (Abdalla et al., 2016). For example, the quantity of maize residue is usually twice that of soybean, but soybean residue decomposes rapidly due to a lower C:N ratio. These, together, can lead to higher SOM with maize residues. Residue retained in NT remains at the soil surface with minimal disturbance, while CT causes some residue incorporation. Consequently, the content of SOM is higher in the uppermost surface 20 cm of NT soils, but it is relatively homogeneously distributed with depth in the surface 20 cm of CT soils (Ziadi et al., 2014). Therefore, the decomposition rate of SOM was largely affected by its distribution in the upper soil layer. As the duration of NT management increases, the contribution of older weathered residue to $CO_2$ emission rises (Oorts et al., 2007). In our analysis, long-term NT with residue retention gave a nearly 26% greater $CO_2$ emission (Table S2.3).

Differences in $CO_2$ and $CH_4$ emissions between the tillage practices became non-significant with time (Figure 2.5a, b). With short-term NT duration, $CO_2$ and $CH_4$ emissions were significantly reduced relative to those with CT, but the differences decreased with longer NT duration. Significantly larger soil carbon stocks in long-term NT made carbon emissions equal to those from the smaller CT soil carbon stocks (Oorts et al., 2007). A new equilibrium, in both tillage systems, between carbon inputs and outputs may have formed. However, in experiments where NT was short-term, the CT and NT soils may not have yet reached the anticipated equilibrium.

Differences in GHG emissions between the tillage practices varied with crop species. Rice production is more likely, among the four crop species evaluated, to exhibit reduced $CO_2$ and $CH_4$ emissions with NT adoption. The surface NT soil bulk density was significantly greater than that of CT soil (Ahmad et al., 2009; Li et al., 2013). Li et al. (2013) speculated that $CH_4$ produced in NT soil might be better retained due to soil surface compaction, thereby making $CH_4$ oxidation by methanotrophic bacteria more likely. Increased bulk density reduces macroporosity, which inhibits organic matter decomposition (Ahmad et al., 2009). This reduces dissolved organic carbon concentration that restricts substrate supply to methanogens and further reduces $CH_4$ production. These NT effects are exclusively significant in paddy rice production because the waterlogged environment otherwise favors $CH_4$ production. Wheat production can enhance SOC and total N sequestration, particularly in NT (Wright et al., 2007), which provides sufficient substrate for $N_2O$ production and possibly explains the larger increase in $N_2O$ emission with NT wheat production.

### 2.5.3 Crop Yield and the GWP of GHG Emissions

In general, our meta-analysis found no significant effect of NT, relative to CT, on crop yield. Previous studies reported a slight yield reduction (about 5%) with NT (van Kessel et al., 2013; Pittelkow et al., 2015). The difference might be because we only analyzed yield data from research trials that included GHG measurements. A great variation was found in the yield dataset. Much of the increased yield in NT was contributed by studies in barley production (Figure 2.5d). Plaza-Bonilla et al. (2014a) reported six-fold greater NT barley yield in a rainfed Mediterranean climate due to better water use efficiency. Pittelkow et al. (2015) further suggested that NT performs better than CT in rainfed conditions in dry climates. Our results concur with these observations. Most (24 out of 25) of the barley yield trials in this meta-analysis were in the dry climate subgroup. Considering there was less of an $N_2O$ emission increase, and a greater decrease in $CO_2$ emission in NT in dry climates (Figure 2.4a, c), NT would be the better management practice for climate change mitigation goals. Our observations that NT increased crop yield by 13.1% in alkaline soils (Figure 2.4d) and by 25.2% at the low N fertilizer rate (Figure 2.5d) were also noteworthy. Additionally, NT exhibited similar GHG emissions to CT in alkaline soils or with low N fertilizer input. These observations suggest that NT can be the better choice under such circumstances.

In terms of climate change mitigation, there is a general consensus that NT can enhance soil carbon sequestration. However, whether this benefit would be offset by NT's stimulation of $N_2O$ emission is still under debate. In this meta-analysis, overall GWP was not different between NT and CT. This suggested that a balance between N2O emission and carbon sequestration under NT can be reached (Halvorson et al., 2008). Moreover, NT induced GWP reductions may not be coincident with yield loss, particularly on acidic soils and with rice production. Considering the other benefits that accompany NT adoption, such as lower labor and machinery inputs, NT may be an effective practice that further mitigates climate change through reduced fossil fuel consumption.

Accordingly, NT implementation can contribute to food security and climate change mitigation. However, interactions between NT and site-specific conditions, including other management practices, could offset NT's benefits. Future NT research should include more field measurements chosen to consider other complicating environment or management factors. For example, measurements of GHG emissions should be for the whole year, as this may better reflect the full expression of emission differences between NT and CT. Due to the limitations in available data, this study focused only on the GHG emissions during the growing season. Emissions during the non-growing season, especially in regions that experience freeze-thaw cycles and snow cover, could be significant and should not be ignored at study sites in these regions. Moreover, analysis of the NT effect from different space and time scales is needed to better identify NT's effectiveness in the context of global change. While meta-analysis is better positioned than individual studies as an effective methodology to generate more informed and accurate conclusions, it depends on data quality and quantity, especially for large scale assessment. The lack of certain meta-data (e.g., fertilization methods, residue management, and soil properties) in some studies made it difficult to include their results in this meta-analysis. Thus, publications should clearly describe weather conditions, site management field operations, and soil properties. In addition, more tillage research regarding all three GHG emissions, measured using similar methods, is needed to draw more representative conclusions regarding tillage choices and resulting GWP.

### 2.5.4 Effects of Conservation Tillage on SOC

Common approaches for enhancing SOC focus on increasing carbon inputs, decreasing losses, or simultaneously affecting both inputs and losses. All conservation tillage practices discussed here, that is, NT and RT, increase soil carbon sequestration to different extents. Previous studies show that conservation tillage increase SOC by only

3%–10% (Luo et al., 2010; Abdalla et al., 2016; Du et al., 2017; Zhao et al., 2017). Our results agree with these earlier findings that conservation tillage increases SOC by 5%. Conservation tillage practices may not necessarily add carbon; their contribution is primarily accomplished by protecting SOC from decomposition and erosion (Six et al., 2000; Lal, 2005). Additionally, conservation tillage can potentially improve soil properties, thereby stimulating more carbon inputs from residue return and rhizodeposition due to promoted plant growth and reducing carbon losses via decreasing leaching and erosion. However, the effectiveness of conservation tillage on SOC sequestration and the mechanisms involved vary with environmental factors and other agronomic practices.

#### 2.5.4.1 Environmental Control in Conservation Tillage

Environmental factors such as climate and soil properties may influence carbon inputs to the soil and affect the processes that regulate carbon loss, considering that all conservation tillage practices are implemented in site-specific climate and soil conditions. The effects on SOC could be affected by environmental factors.

*Climatic variability:* Climate is one of the major driving forces that regulate SOC distribution. On average, SOC accumulation is greater than decomposition in cool, wet areas than in dry, warm regions (Jobbágy and Jackson, 2000). Soil carbon is positively related to precipitation and negatively correlated with temperature (Rusco et al., 2001), with the former correlation tending to be stronger (Martin et al., 2011; Meersmans et al., 2011). High precipitation is usually associated with abundant growth and high rates of carbon inputs to soils (Luo et al., 2017), while low temperatures may remarkably reduce microbial activity, resulting in low rates of organic matter decomposition and measurable amounts of SOC accumulation (Castro et al., 1995; Garcia et al., 2018). No-tillage increased SOC with no significant difference between aridity conditions (Table 2.3), although NT performed better at storing SOC in arid areas (Figure 2.9a). This result suggests that arid-region soils have a high potential to store carbon when using proper management practices (Tondoh et al., 2016). In addition, NT can enhance carbon sequestration more in warm areas than in cool areas. The temperature could affect the establishment and growth of cover crops (Akemo et al., 2000). In warm areas, cover crops may develop well and potentially capture more carbon dioxide ($CO_2$) from the atmosphere, thus providing more carbon inputs into soils after they die (e.g., Bayer et al., 2009). Tillage results in the breakdown of macroaggregates and the release of aggregate-protected SOC (Six et al., 2000; Mikha and Rice, 2004). Tillage-induced SOC decomposition usually proceeds at higher rates in warm than in cool areas. Implementing NT, with minimal soil disturbance, protects SOC from decomposition. As a result, SOC increases can be more significant in warm conditions considering the relatively higher baseline of the decomposition rate compared to that in cool areas.

*Soil properties:* Soil organic carbon is strongly correlated with clay content, with an increasing trend toward more SOC in fine-textured soils (Stronkhorst and Venter, 2008; Meersmans et al., 2012). The SOC mineralization rate probably diminishes as clay concentrations increase (Sainju et al., 2002). Clay minerals can stabilize SOC against microbial attacks through the absorption of organic molecules (Ladd et al., 1996). By binding organic matter, clay particles help form and stabilize soil aggregates, imposing a physical barrier between decomposer microflora and organic substrates and limiting water and oxygen available for decomposition (Dominy et al., 2002). The ability of conservation tillage to enhance SOC differs with soil texture (Figure 2.10a). Considering conservation tillage merely reduces soil disturbance and normally does not add extra materials to soils, it can be inferred that the effect of conservation tillage on SOC is texture dependent.

Soil depth may potentially influence the effects of conservation tillage on SOC (Baker et al., 2007). Conservation tillage was most beneficial to SOC accumulation in surface soils. For example, NT increased SOC by 7% in the 0–3 cm soil layer (Abdalla et al., 2016) and by 3% at the 40 cm depth (Luo et al., 2010). Our findings suggested that conservation tillage can enhance SOC sequestration in the entire soil profile, although the positive effects vary with soil depths (Table S2.5). Conventional tillage breaks soil aggregates and increases aeration and thus enhances soil organic matter mineralization (Cambardella and Elliott, 1993). Conventional tillage also incorporates residues into deeper soil layers, resulting in a more uniform distribution of SOC (albeit at lower concentrations) in the soil profile (Sainju et al., 2006; Plaza-Bonilla et al., 2010). In contrast, conservation tillage keeps residues at the soil surface and reduces their degree of incorporation into the soil (Franzluebbers et al., 1995). Nevertheless, the positive effects of NT on SOC have been found in a deep soil profile (0–60 cm, Liu et al., 2014).

Soil pH is recognized as a dominant factor governing the soil organic matter turnover rate, although its mode of impact is still unclear (Van Bergen et al., 1998). Soil pH affects selective presentation or metabolic modification of specific components (e.g., lignin-cellulose, lipids) during decomposition (Kemmitt et al., 2006) and, therefore, abiotic factors (e.g., carbon and nutrient availability) and biotic factors (e.g., the composition of the microbial community). Also, soil pH can change the decomposition rate of crop residues and SOC via its effect on SOC solubility and indirectly by altering microbial growth, activity, and community structure (Pietri and Brookes, 2009; Wang et al., 2017).

The levels of soluble organic carbon may increase with increasing acidity (Willett et al., 2004; Kemmitt et al., 2006). Motavalli et al. (1995) suggested that increased soil acidity would cause greater soil organic matter accumulation due to reduced microbial mineralization; however, this was challenged by Kemmitt et al. (2006), who found no significant trend in SOC in response to pH changes. In this study, conservation tillage resulted in greater increases in SOC in neutral or alkaline soils compared to acid soils.

Table 2.3 Between-group variability ($Q_M$) of the variables controlling the effects of no-tillage and reduced tillage on soil organic carbon

| Variables | No-till | | Reduced tillage | |
|---|---|---|---|---|
| | df | $Q_M$ | df | $Q_M$ |
| Duration | 2 | 12.14** | 2 | 13.69** |
| Aridity index | 1 | 0.13 | 1 | 10.99*** |
| Mean annual air temperature | 1 | 16.32*** | 1 | 0.47 |
| Soil texture | 5 | 20.98*** | 5 | 32.15*** |
| Soil depth | 3 | 210.69*** | 3 | 73.38*** |
| Soil pH | 2 | 9.8** | 2 | 3.52 |
| Residue | 1 | 6.56* | 1 | 0.04 |
| Nitrogen fertilization | 3 | 7.62 | 3 | 11.43* |
| Irrigation | 1 | 9.61** | 1 | 0.92 |
| Crop rotation | 1 | 1.72 | 1 | 0.26 |

Statistical significance of $Q_M$: * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

#### 2.5.4.2 Conservation Tillage and Other Agronomic Practices

Crop residues provide substantial amounts of organic matter and may influence the effect of conservation tillage on SOC. Residue retention changes the formation of soil macroaggregates (Benbi and Senapati, 2010), promoting SOC preservation and accumulation (Six et al., 2002). Residue cover protects the soil surface from direct impact by raindrops (Blanco-Canqui et al., 2014). In addition, crop residues provide organic substrates to soil microorganisms that can produce binding agents and promote soil aggregation (Guggenberger et al., 1999). Conversely, residue removal reduces carbon input to the soil system and ultimately decreases SOC storage (Manna et al., 2005; Koga and Tsuji, 2009). This suggests that the amount of carbon inputs predominantly controls changes in SOC stocks (Virto et al., 2012). For NT, enhancing SOC was significantly greater with residue return than with residue removal. Our study suggests that changes in SOC did not differ with residue management in RT (Table 2.3), although a slightly greater increase in SOC occurred with residue retention than with residue removal (Figure 2.11b). This unexpected result is likely due to the limited number of observations with residue removal. Another possible reason is that the interaction between residue management and soil type may lead to various responses in SOC stocks. For example, residue removal increased SOC by 3.6%, while residue retention had no effect on SOC in clay and clay loam soils. The decomposition of crop residues involves complex processes, which are controlled by multiple biogeochemical and biophysical conditions.

Nitrogen fertilization noticeably increases SOC stock but with diminishing returns. For example, Blanco-Canqui et al. (2014) indicate that nitrogen fertilizer increases SOC when the nitrogen fertilization rate is below 80 kg N $ha^{-1}$, above which it reduces aggregation and then decreases SOC stocks. Nitrogen fertilization can stimulate biological activity by altering carbon/nitrogen ratios, thereby promoting soil respiration and decreasing SOC content (Mulvaney et al., 2009); however, excessive nitrogen addition may reduce soil fungi populations, inhibit soil enzyme activity, and decrease $CO_2$ emissions (Wilson and Al Kazi, 2008). These findings suggest that nitrogen fertilization enhances the positive effect of conservation tillage on SOC, likely through increased plant biomass production (Gregorich et al., 1996).

Aridity can limit plant growth and crop residue return and ultimately compromise SOC accumulation (Moreno et al., 2006). Conservation tillage can potentially enhance soil water retention by improving soil porosity and erosion control. Irrigation ensures sufficient water for plant growth, resulting in more biomass production than in rainfed conditions (Shipitalo et al., 1990; Chan, 2004; Capowiez et al., 2009; Swanepoel et al., 2016). The crop root density is much higher in irrigated conditions compared to rainfed conditions (Jobbágy and Jackson, 2000), leading to higher organic matter input. Thus, CSA management practices, in combination with irrigation, could further increase SOC content.

Rotational cropping potentially provides high carbon input to soils. Compared to continuous cropping systems, crops in rotational cropping systems have a greater belowground allocation of biomass (Van Eerd et al., 2014), resulting in more inputs of crop residue to the soil system. Enhancing rotation complexity can benefit carbon sequestration (West and Post, 2002). The present analysis suggests that conservation tillage can increase SOC sequestration regardless of the crop rotation system.

Conservation tillage, together with cover crops, may enhance SOC storage more. For example, in sandy clay loam and loamy sand soils, the sum of the effect size was 21% and 31%, respectively. Coarse-textured soils are not carbon-saturated and have great potential for carbon uptake. Cultivated land tends to suffer from SOC degradation, and SOC accumulation could quickly increase upon initiating farming practices due to high carbon inputs to the soil system (Vieira et al., 2009). For example, in sandy loam soils, Higashi et al. (2014) showed that SOC increased by 22% with a combination of cover crops and NT. These results may be attributed to the stability of soil water-stable aggregates when cover crops are grown in sandy clay loam soils (McVay et al., 1989), given that aggregate stability has been linked to the protection of SOC from mineralization (Unger, 1997). The combination of cover crops and conservation tillage significantly decreased SOC in clay soils. The reason for this unexpected result may be due to the limited number of study sites where this combination of treatments was evaluated (a few data pairs in our meta-analysis), and the burning of cover crop biomass (Tian et al., 2005).

## 2.6 Conclusion

This study provided a comprehensive and quantitative synthesis of NT and CT effects on GHG emissions and crop yield in different cropping systems. In general, NT can reduce $CH_4$ emission by 15.5%, with a concomitant increase in $N_2O$ emission of 10.4%. These effects seem to diminish with the long-term duration of NT. Thus, the combination of NT with other climate-smart agriculture (CSA) components might be needed. Although NT cannot reduce all three GHG emissions simultaneously, there is some evidence of a reduction in overall GWP in NT given specific conditions, which needs to be verified with further observations. The emission of $CO_2$ can be significantly reduced, with a yield benefit, with NT adoption in dry climates. However, in humid climates, NT tended to increase $N_2O$ emission and reduce crop yield, suggesting careful consideration of NT adoption in humid regions. Soil pH was also important, and implementing NT can help mitigate climate change on acidic soils and enhance food security on alkaline soils because total GWP was reduced in NT without yield penalty on acidic soils, and NT increased crop yield without affecting GWP on alkaline soils. No-tillage and a low N fertilizer rate increased crop yield without exacerbating GHG emissions compared to CT. Furthermore, subsurface N fertilizer placement in NT should be considered to reduce $N_2O$ emission. Among the four cereal crops, there was a large reduction in GWP with NT rice production, with no yield penalty. A yield benefit was only observed for barley, while there was a maize yield loss. These results credit NT for enhancing climate change mitigation and food security in rice and barley production, respectively. Overall, this study provides both support and caution to the adoption of NT as a CSA management practice. To identify other CSA management practices that are suitable at local, regional and/or global scales, future CSA research programs that systematically investigate agroecosystem responses (e.g., crop yield, GHG emissions) using diverse methods (e.g., observation, meta-analysis, and agroecosystem modeling) are needed.

Based on 2,180-paired comparisons from 297 peer-reviewed articles, our meta-analysis quantitatively analyzed SOC changes as influenced by conservation tillage and associated environmental factors and other agronomic practices. Although our results present the positive effects of conservation tillage on soil carbon storage, it may have constraints regarding the ability to enhance soil carbon sequestration. The SOC benefit of conservation tillage strongly depends on environmental factors and other agronomic practices. Therefore, the choice of proper practices is potentially highly region-specific. Our results imply that conservation tillage has great potential for climate change mitigation when combined with cover crops.

# CHAPTER 3. ASSESSING THE LONG-TERM EFFECTS OF NO-TILLAGE ON SOIL CARBON DYNAMICS IN A MAIZE-COVER CROP SYSTEM[2]

## 3.1 Abstract

Climate-smart agriculture management practices such as no-tillage (NT) and cover crops (CCs) have been widely applied and are expected to offer multiple environmental benefits (e.g., soil carbon sequestration, yield stability, and climate resilience). However, the long-term effects of these management practices, especially their synergistic interaction, have not been well addressed. This study used an improved agroecosystem model (DLEM-Ag) to explore the synergistic effects of NT and CCs on soil carbon dynamics in a continuous maize system in the middle south of the US for 1970-2099. Simulation results for 1970-2018 show that NT, relative to conventional tillage (CT), led to carbon gains (0.22 Mg C $ha^{-1}$ $yr^{-1}$) in the topsoil in a CC-inclusive continuing maize system; however, NT *per se* brought minor net carbon gains. This well captures the field observations. Model factorial analyses reveal that soil carbon sequestration was highly correlated with biomass carbon inputs from both the winter cereal CC and the summer maize. Elevated $CO_2$ and warming effects were the main contributors to soil carbon gains, as these promote CC growth. Further model projections suggest that soil organic carbon would increase in the RCP 8.5 future scenarios (2019-2099), with greater gains under NT-CCs than under CT-CCs (0.089 vs. 0.058 Mg C $ha^{-1}$ $yr^{-1}$), largely due to enhanced CC biomass production. Moreover, NT-CCs would reduce carbon loss compared to CT-CCs

[2] Based on Huang, Yawen, et al. (2020) “Assessing synergistic effects of no-tillage and cover crops on soil carbon dynamics in a long-term maize cropping system under climate change” *Agricultural and Forest Meteorology*, 291, 108090.

(-0.002 vs. -0.017 Mg C $ha^{-1}$ $yr^{-1}$) in the RCP 2.6 scenarios. Our study highlights the importance of CCs in enhancing cropland carbon sequestration and indicates that NT and CCs, taken together, can serve as a viable strategy to ensure crop production through promoting soil health in similar maize cropping systems.

## 3.2 Introduction

Soil organic carbon (SOC) is a key soil health indicator and plays a crucial role in providing soil ecosystem services (Lorenz and Lal, 2016). Estimating SOC changes with climate change has been receiving considerable attention because these changes significantly affect food production and soil biogeochemical cycles and drive climate change feedbacks by altering soil quality and accumulation rates of atmospheric $CO_2$ (Lal, 2014). Moreover, for both agronomic and environmental purposes, maintaining or increasing SOC stocks in agricultural soils represents an essential component of sustainable land management.

Conservation tillage and cover crops are among the most popular management practices to mitigate soil erosion and degradation (Blanco-Canqui et al., 2015; Townsend et al., 2016; Kaye and Quemada, 2017). They are also recognized as two widely used climate-smart agriculture practices that aim to enhance food security and build resilience to climate change (Lipper et al., 2014; Bai et al., 2019). In general, conservation tillage alleviates soil disturbance and maintains soil surface residue cover, which helps improve soil aggregation and stability (He et al., 2011), conserve soil water (Plaza-Bonilla et al., 2014), and reduce soil erosion (Puget and Lal, 2005). Cover crops have the potential to reduce erosion and nitrogen leaching and improve soil health (Blanco-Canqui et al., 2015; Poeplau and Don, 2015; Liu et al., 2019).

Many analyses have been conducted to assess the potential of these practices to improve SOC stocks. For example, SOC sequestration rates due to no-tillage (NT), compared to conventional tillage (CT), were estimated to be 0.40 ± 0.61 Mg C $ha^{-1}$ $yr^{-1}$ (n = 44) in the central USA (Johnson et al., 2005) and 0.45 ± 0.04 Mg C $ha^{-1}$ $yr^{-1}$ (n = 147) in the southeastern USA (Franzluebbers, 2010). Some recent meta-analyses have argued that NT often redistributes carbon nearer to the soil surface but does not increase SOC stocks as compared with CT (Olson et al., 2014; Powlson et al., 2014), whereas Bai et al. (2019) suggested that NT can enhance SOC sequestration in the whole soil profile, possibly due to greater root growth. The significant discrepancies in reported results have not been well addressed and could be attributed to experiment duration, soil sampling frequency and depth, and the analytical approach to SOC determination. Other factors, such as climate and soil properties interacting with management factors (e.g., crop rotation, cover crops, nitrogen, and drainage; Luo et al., 2010; Ugarte et al., 2014; Bai et al., 2019) have shown synergistic effects with tillage on SOC sequestration. Including cover crops in NT systems might lead to more SOC accumulation (Blanco-Canqui et al., 2013; Higashi et al., 2014; Bai et al., 2019). However, a knowledge gap still exists regarding the underlying mechanisms responsible for enhanced soil carbon sequestration. Information regarding the synergistic effects of conservation tillage and cover crops on soil carbon dynamics (such as various soil pools and fluxes) is also far from certain. Moreover, most reported results were based on relatively short-term field experiments (usually <10 years), and limited environmental control experiments. However, building and maintaining SOC stocks requires a sustained and consistent effort and the long-term examination of SOC stock variation, as influenced by multiple management practices as well as associated underlying mechanisms in the context of climate change, is crucial.

Recently, some crop (Basche et al., 2016; Iocola et al., 2017) and soil carbon (Maas et al., 2017; Nash et al., 2018) models have been used to evaluate the individual effects of tillage and cover crops on crop yield and SOC, but few of them addressed interactive effects of tillage and cover crops. Besides, most ecosystem models that are used to investigate terrestrial biogeochemical cycles have not included detailed representations of tillage practices (Lutz et al., 2019). These limitations might bring large uncertainties to estimating the role of agriculture in the global biogeochemical balance and in assessments to strengthening resilience to climate change.

In this study, we applied an improved process-based agroecosystem model (DLEM-Ag), in which conservation tillage and cover crops were represented, to examine the long-term synergistic effects of NT and cover crops on soil carbon dynamics. We conducted a range of simulation experiments for a long-term continuous maize cropping system in the middle south of the US, where the NT and cover crop (NT-CC) were applied during the 1970-2018 period. The overarching objectives were to: (1) evaluate model performance against observed crop yield and SOC in the NT-CC system; (2) explore the underlying mechanisms responsible for synergistic effects of NT-CC on long-term soil carbon dynamics; (3) attribute long-term variation in SOC to major influencing factors during the historical period; and (4) perform trajectory prediction under future climate scenarios.

## 3.3 Materials and Methods

### 3.3.1 Site Description

The field experiment was established in 1970 at the Kentucky Agricultural Experiment Station farm ("Spindletop") near Lexington, KY, USA (N 38°07′24″, W 84°29′50″) with two tillage systems (moldboard plowing with secondary tillage, which is considered conventional tillage [CT]; no-tillage [NT]) and four mineral nitrogen application rates (0, 84, 168, and 336 kg N $ha^{-1}$ $yr^{-1}$). The experiment was laid out in a split-block design with four replications. The soil is a moderately weathered, well-drained Maury silt loam (fine, mixed, semiactive, mesic Typic Paleudalf) on a 1 to 3% slope without evident rill erosion. Before establishing the experiment in 1970, the site had been a bluegrass (*Poa pratensis* L.) pasture for about 50 yr (Frye and Blevins, 1996). The experimental site is characterized by a rainfed moderate humid climate with a mean annual temperature of 13.1°C and mean annual precipitation of 1222 mm, though with slightly increasing trends of 0.03°C $yr^{-1}$ and 3.8 mm $yr^{-1}$ from 1970 to 2018 (Figure 3.1a, c respectively). Additional details about the site, sampling, and analysis not included in this paper are available in previous studies (Blevins et al., 1971; Blevins et al., 1977; Grove and Blevins, 1988; Ismail et al., 1994).

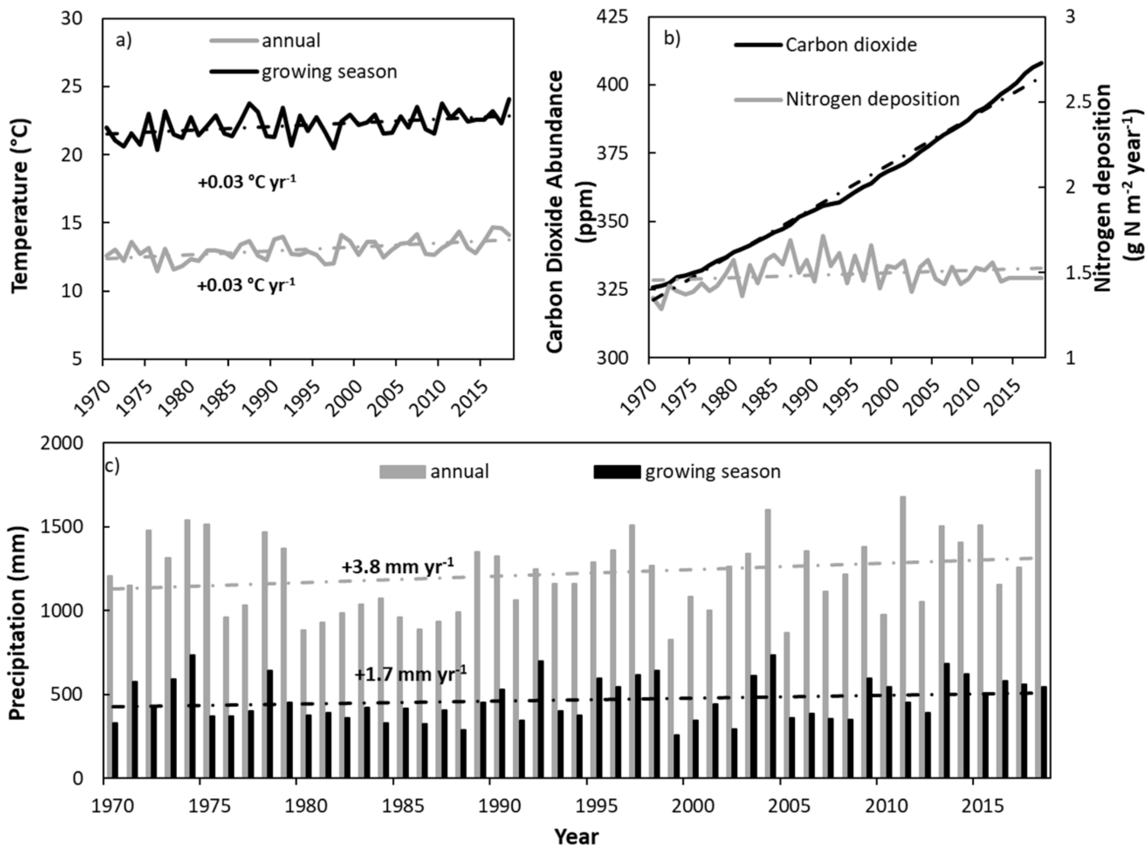


Figure 3.1 Annual and growing season average temperature (a), annual atmospheric $CO_2$ concentration, and nitrogen deposition (b), annual and growing season total precipitation (c) over Lexington from 1970 to 2018. Dashed lines represent the linear trends

In the experiment, maize (Zea mays L.) was grown each year, followed by a winter cereal cover crop. Tilled plots were plowed and disked in mid-April, about 1-2 weeks before maize planting in early to mid-May. The average plowing depth was about 20 cm. Within one week after planting, ammonium nitrate was broadcast over the soil surface. Following harvest, maize residues were left on the soil surface. Previous work at this site has demonstrated that the consistent nitrogen-sufficient fertilizer rate for maximizing crop yields was 168 kg N $ha^{-1}$ (Grove et al., 2009). In this study, we only considered the nitrogen-sufficient tillage treatments (i.e., NT-168 and CT-168), to exclude the influence of nitrogen fertilizer on the outcomes.

### 3.3.2 Datasets

*Site data collection*: Soil samples were taken to varying depths in varying increments over the years. Soil data usable for this study were collected about every eight to nine years (Blevins et al., 1977; Blevins et al., 1983; Ismail et al., 1994; Grove et al., 2009). Specifically, samples taken in 1975, 1980, 1985, and 1989 were from all plots at depths of 0 to 5, 5 to 15, and 15 to 30 cm (Blevins et al., 1977; Blevins et al., 1983; Ismail et al., 1994). Samples in 2008 were taken to a depth of 100 cm in 10 cm increments from plots receiving 0, 168, and 336 kg N $ha^{-1}$ (Grove et al., 2009). In this study, we took samples from all plots to a depth of 30 cm in 2018. Sampling events occurred either in the spring pre-plant or in the fall post-harvest. SOC for the initial year (1970) is from the bluegrass pasture surrounding the experiment plots. In addition, SOC was determined by the Walkley-Black method before 1989 and by dry combustion thereafter. The data within each treatment were averaged to the depth of 30 cm each year. Crop yields were available from 1970 to 1990 (Ismail et al., 1994) and from 2013 to 2017. A carbon content factor (450 g C per Kg) was applied to convert the dry biomass to carbon (Prince et al., 2001). We also measured $CO_2$ fluxes bi-weekly during the 2018 growing season using an FTIR-based field gas analyzer (Gasmet DX4040, Gas Technologies Oy, Helsinki, Finland), with the static chamber method (Parkin and Venterea, 2010). The gas fluxes were calculated following the method of Iqbal et al. (2013).

*Model input data*: To drive the model simulations, we collected historical climate data from the National Climate Data Center for the Lexington, KY, USA weather station. Historical $CO_2$ and nitrogen deposition datasets were obtained from the Earth System Research Laboratory of NOAA (National Oceanic and Atmospheric Administration, https://www.climate.gov/) and ISIMIP (Inter-sectoral Impact Model Intercomparison Project, https://esg.pik-potsdam.de/projects/isimip/), respectively. Atmospheric $CO_2$ concentration increased from 325 ppm in 1970 to 408 ppm in 2018, whereas there was no significant trend in nitrogen deposition over the study region (Figure 3.1b).

Future climate scenarios were obtained from the Downscaled CMIP3 and CMIP5 Climate and Hydrology Projections (https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/). We utilized 36 CMIP5 global climate model (GCM) outputs (Table S3.1) under the representative carbon pathway (RCP) 2.6 and 8.5 scenarios to drive the model simulations through 2099. The RCP 2.6 represents a low emission scenario with significant climate action, aiming to limit the increase in global mean temperature to less than 2°C by 2100. The RCP 8.5 represents the business-as-usual high emission scenario, yielding a range of temperature outcomes of +4.0 to 6.1°C by 2100 (IPCC, 2014). The corresponding RCP $CO_2$ data was obtained from the recommended RCP CMIP5 datasets (Meinshausen et al., 2011), in which the projected atmospheric $CO_2$ concentrations would be 421 ppm and 927 ppm by 2099 under RCP 2.6 and RCP 8.5 scenarios, respectively.

### 3.3.3 Model Description

The agricultural version of the Dynamic Land Ecosystem Model (DLEM-Ag) is a highly integrated process-based agroecosystem model which simulates: (1) the daily crop growth and exchanges of trace gases ($CO_2$, $CH_4$, and $N_2O$) between agroecosystems and the atmosphere; and (2) fluxes and storage of carbon, water, and nitrogen within agroecosystem components that are affected by multiple factors like climate, atmospheric $CO_2$, nitrogen deposition, tropospheric ozone, land use and land cover change, and agriculture management practices (e.g., harvest, rotation, irrigation, and fertilizer use). The DLEM-Ag has been extensively used to study crop production, SOC, and exchanges of trace gases between agroecosystems and the atmosphere. The detailed structure and processes have been well documented in previous work (e.g., Tian et al., 2010; Ren et al., 2011; Ren et al., 2012; Ren et al., 2016; Zhang et al., 2018). Here, we provide a brief introduction to the modules for plant growth and soil carbon-related processes, as well as the tillage module.

#### 3.3.3.1 Crop productivity

The crop gross primary productivity (GPP, g C $m^{-2}$ $day^{-1}$) in the DLEM-Ag is calculated by scaling leaf assimilation rates (μmol $CO_2$ $m^{-2}$ $s^{-1}$) up to the whole canopy (Farquhar et al., 1980). The canopy is divided into sunlit and shaded layers.

$$GPP_a = GPP_o f(PAR) f(T) f(P) f(CO_2) f(O_3) f(N) \tag{3.1}$$

where $GPP_a$ is the actual GPP derived from the potential GPP ($GPP_o$) under optimized conditions adjusted by environmental factors that directly or indirectly influence carbon assimilation and allocation, including photosynthetic active radiation $f(PAR)$, air temperature $f(T)$, precipitation $f(P)$, atmospheric $CO_2$ concentration $f(CO_2)$, tropospheric $O_3$ $f(O_3)$, and nitrogen availability $f(N)$ associated with nitrogen deposition and fertilization. More detailed information is provided in Text S3.1.

#### 3.3.3.2 Soil Carbon Processes

In DLEM-Ag, soil organic matter consists of six soil carbon pools (i.e., three microbial pools, two slow organic matter pools, and one dissolved organic matter pool), plus two woody debris pools (above- and belowground woody debris), and four litter pools (above- and belowground, easy and resistant to decomposition). The size of each pool and the carbon fluxes transferred between pools determine the source and loss of soil organic and inorganic carbon. Generally, all carbon inputs from tissue turnover and crop residue are allocated to litter pools according to the carbon/nitrogen ratio. Then the carbon fluxes are transferred between pools through biological decomposition, physical adsorption, desorption, surface runoff, and leaching. The decomposition rate of each pool is estimated using a first-order algorithm (Parton et al., 1994) that is influenced by soil temperature, water content, nutrient availability, and soil texture. Details can be found in previous studies (e.g., Banger et al., 2015; Ren et al., 2012, 2020; Tian et al., 2015).

#### 3.3.3.3 Tillage

In this study, we improved the DLEM-Ag by incorporating a tillage sub-module to specifically examine the effects of NT on soil carbon balance in the agroecosystem (Figure 3.2). The representation of cover crops in the model is simply to consider the growth of winter cereal rye, with all its biomass left in the field after termination. The primary input of organic carbon to the soil is vegetal, and SOC losses occur mainly through decomposition, runoff, and leaching. The incoming carbon at this site consists of three major pathways, i.e., growing season litterfall, dead cover crop materials, and maize residue (including shoots and roots) return after harvest. The outgoing carbon pathways include $CO_2$ emissions, runoff, and leaching, as well as others, such as $CH_4$ emissions and volatile organic compounds. In this study, we defined leaching carbon as the sum of particle organic carbon loss through surface runoff and dissolved organic carbon loss through subsurface drainage.

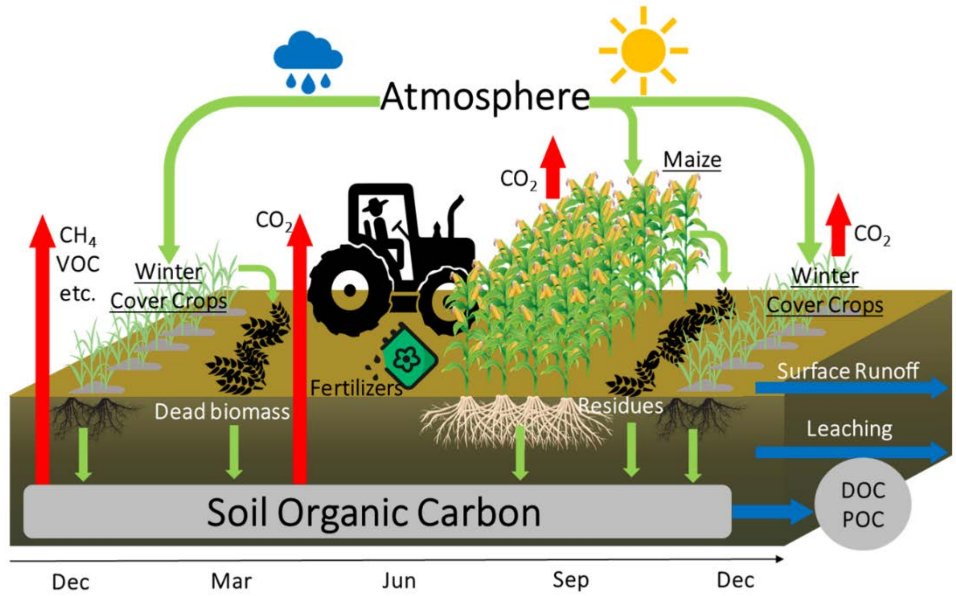


Figure 3.2 A diagram illustrating the carbon cycle at the experiment site (green arrows denote carbon input pathways, red arrows for carbon output pathways, and blue arrows for lateral carbon fluxes)

The tillage sub-module in DLEM-Ag mainly considers mixing effects within the tilled soil layer and subsequent effects on soil water processes, as well as the direct effects on soil organic matter decomposition rates.

***Tillage mixing***. The tillage operation redistributes residue and nutrients in the tilled soil layer, with a mixing efficiency, $f_{mix}$ (Table 3.1, adapted from Buckingham and Paul, 1993), which depends on the tillage practice and defines the fraction of a residue/nutrient/SOC pool in each soil layer that is redistributed through the depth of soil that is mixed by the operation (Williams et al., 2012);

$$X(L) = X_0 \times (1 - f_{mix}) + SMX_0 \times f_{mix} \times \frac{Z(L)-Z(L-1)}{D_t} \quad (3.2)$$

where $X$ is the amount of carbon or nitrogen in each pool in layer $L$ after mixing, and $X_0$ is the original amount before mixing. The parameter $D_t$ is the tillage depth, $SMX_0$ is the sum of $X_0$ in $D_t$, and $Z$ is the depth to the bottom of the tilled layer.

***Residue coverage***. Litter pools in the DLEM-Ag generally consist of two main components ($Litter_{ag}$ and $Litter_{bg}$, the aboveground and belowground litter pools, respectively). Crop residues left on the field are transferred to $Litter_{ag}$. Root residues are transferred to $Litter_{bg}$. The fraction of the soil surface covered by the remaining residue ($Res_{fc}$) is calculated by adapting the equation from Gregory (1982):

$$Res_{fc} = 1.0 - e^{-AM \times Litter_{ag}} \tag{3.3}$$

where $AM$ is the area covered per unit dry weight of residue (ha kg$^{-1}$, Table 3.1) and depends on residue type (e.g., crop, density). The $AM$ values in this study are adapted from Dadoun (1993). A fraction of residue from $Litter_{ag}$ is transferred to $Litter_{bg}$ due to the mixing effect of tillage. The total thickness of surface residue ($Res_{thick}$) is estimated following Dadoun (1993):

$$Res_{thick} = \sum_{i=1}^{n} S_i \times A_{thick} \tag{3.4}$$

$$S_i = S_{i-1} \times \left(1.0 - e^{\left(-AM * \frac{M_i}{S_{i-1}}\right)}\right) \tag{3.5}$$

$$M_i = M_{i-1} - S_{i-1}/AM \tag{3.6}$$

The algorithm assumes that the residues are arranged in layers with the coverage of layer by equation (3.3). The mass of residue ($M_i$, kg/ha) overlying an adjacent lower layer $(i-1)$ is the difference between the overlying biomass from the previous calculation step ($M_{i-1}$) and the biomass needed to cover the underlying residue layer ($S_{i-1}$, ha residue per ha ground surface). $A_{thick}$ is the average thickness of a residue layer with 100% coverage (1.5 cm), and $n$ is the number of layers. These calculations are iterated until no surface residues are left.

***Rainfall interception***. Crop residues intercept a significant amount of rainfall. The maximum residue water storage capacity ($S_{res\_max}$) is a crop-specific parameter. According to Kozak et al. (2007), a quadratic equation was fitted to the maximum interception amounts giving:

$$S_{res\_max} = aM_{res} - bM_{res}^{2} \tag{3.7}$$

where $M_{res}$ is the residue mass, and $a$ and $b$ are crop-specific parameters (Table 3.1, Kozak et al., 2007). The amount of precipitation intercepted is a balance of the amount of water currently held ($S_{res}$) and the maximum residue storage. All the $S_{res}$ is assumed to be available for evaporation.

***Evaporation***. The effect of surface residues on evaporation is adapted from Andales (1998). The energy available for soil evaporation, i.e., soil potential evaporation ($PE_{soil}$) is budgeted for two processes; evaporation of water contained in the residues and evaporation of water contained in the soil. For evaporation of water in the residues, the $PE_{soil}$ is decreased by the amount of water evaporating from the residues, and the residue water content is updated:

$$\begin{cases} if\ PE_{soil_i} < S_{res}: S_{res_i} = S_{res_f} - PE_{soil_i}\ and\ PE_{soil_f} = 0 \\ if\ PE_{soil_i} \geq S_{res}: S_{res_f} = 0\ and\ PE_{soil_f} = PE_{soil_i} - S_{res_i} \end{cases} \tag{3.8}$$

where the subscripts *i* and *f* designate initial and final values (before and after evaporation from residues), respectively. For evaporation from the soil, surface residues that serve as a physical barrier further reduce soil potential evaporation. The following function is used to calculate the decrease in $PE_{soil}$ from a surface partially covered by residues relative to bare soil ($R_{cov}$, Dadoun, 1993):

$$R_{cov} = 1 - 0.807 \times Res_{fc} \tag{3.9}$$

For higher residue loads that provide a full cover, the thickness of the residue layer is used to predict the relative decrease in soil evaporation ($R_{thick}$, Dadoun, 1993):

$$R_{thick} = e^{-0.5 \times Res_{thick}} \tag{3.10}$$

The reduced soil potential evaporation is:

$$PE_{soil\_f} = \min\left(R_{cov}, R_{thick}\right) \times PE_{soil\_i} \tag{3.11}$$

where the subscripts $i$ and $f$ indicate values before and after reduction due to residue barriers, respectively.

Effects on decomposition. In DLEM-Ag, the decomposition rate for each soil carbon pool ($k_{pool}$) is influenced by soil temperature, water content, nutrient availability, and texture:

$$k_{pool} = kmax_{pool} \times f(T) \times f(W) \times f(N) \times f(clay) \tag{3.12}$$

$$f(T) = 4.89 \times e^{-3.432+0.1\times T\times(1-0.5\times T/36.9)} \tag{3.13}$$

$$f(W) = \begin{cases} \dfrac{1-e^{-\theta/\theta_{sat}}}{1-e^{-\theta_{fc}/\theta_{sat}}} & \theta \le \theta_{fc} \\ 1.0044 - \dfrac{0.0044}{e^{-5\frac{\theta/\theta_{sat}-\theta_{fc}/\theta_{sat}}{1-\theta_{fc}/\theta_{sat}}}} & \theta > \theta_{fc} \end{cases} \tag{3.14}$$

$$f(clay) = 1 - 0.75P_{clay}/100 \tag{3.15}$$

$$f(N_{mi}) = \begin{cases} 1 - \dfrac{avn-avn_{opt}}{avn_{opt}} & avn > avn_{opt} \\ 1 & avn_{opt}/2 \le avn \le avn_{opt} \\ 1 + \dfrac{0.5avn_{opt}-avn}{avn_{opt}} & avn \le avn_{opt}/2 \end{cases} \tag{3.16}$$

$$f(N_{im}) = avn/n_{imm} \tag{3.17}$$

where $k_{pool}$ is the actual decomposition rate for each pool derived from the potential decomposition rate ( $kmax_{pool}$) under optimized conditions, adjusted by environmental factor scalars, including soil temperature $f(T)$, soil moisture $f(W)$, soil nitrogen $f(N)$, and soil texture $f(clay)$. The $f(N_{mi})$ and $f(N_{im})$ are different calculations of $f(N)$ due to mineralization and immobilization, respectively. $\theta$ is soil water content (mm); $T$ is soil temperature (°C); $\theta_{sat}$ is soil water content at saturation (mm); $\theta_{fc}$ is soil water content at field capacity (mm); $P_{clay}$ is the percentage of clay content (%); $avn$ is the available soil nitrogen (g N/m$^2$); $avn_{opt}$ is the optimum available soil nitrogen (g N/m$^2$); $n_{imm}$ is the potential nitrogen immobilization estimated by the tentative decomposition procedure. A fraction of the decomposed carbon from each pool is converted to $CO_2$ through heterotrophic respiration, and the left is transferred to other pools.

In the tillage module, tillage directly affects soil organic matter decomposition rates within the tilled depth. Once a tillage operation is executed, a tillage scalar is added to the equation (3.12). Then,

$$k_{pool} = kmax_{pool} \times f(T) \times f(W) \times f(N) \times f(clay) \times f_{till} \tag{3.18}$$

According to Neitsch et al., (2011), tillage can enhance the decomposition rate through the factor $f_{till}$. This factor is calculated independently for each soil layer and depends on the tillage mixing efficiency ($f_{mix}$) and the soil texture. The $f_{till}$ basal value is 1, and it is enhanced immediately after a tillage event based on the estimated cumulative $f_{mix}$ (or $f_{cm}$):

$$f_{till,i} = 1 + f_{cm,i} \tag{3.19}$$

$$f_{cm,1} = (3 + 5e^{-5.5clay})\left(\frac{f_{mix}}{f_{mix}+e^{1-2f_{mix}}}\right) \tag{3.20}$$

$$f_{cm,i} = f_{cm,i-1} \times \left(1 - 0.02 \times \frac{\theta}{\theta_{sat}}\right),\ \ i > 1 \tag{3.21}$$

where θ and $\theta_{sat}$ are the current and saturated soil moisture contents of a given layer at day i. The factor $f_{till}$is reduced daily based on soil moisture (Eq. 3.21), to simulate soil settling. If $f_{till}$> 1 and a tillage operation is executed, the corresponding $f_{mix}$ must be added to the current $f_{cm}$.

Table 3.1 List of parameters for tillage submodules to drive the DLEM-Ag model

| Operation | Mixing efficiency of tillage (0-1) | Mixing depth of tillage (cm) | Litter coverage ($m^2$ $g^{-1}$) | | Litter interception coefficient ($10^{-4}$ mm ha $kg^{-1}$) | |
|---|---|---|---|---|---|---|
| | | | Maize | Winter rye | Maize | Winter rye |
| NT | 0.1 | 5 | 0.004 | 0.005 | 3.46, 1.05 | 3.55, 0 |
| CT | 1 | 20 | 0.004 | 0.005 | 3.46, 1.05 | 3.55, 0 |

### 3.3.4 Model Calibration and Evaluation

The simulated results were validated against available data for grain yield, SOC, and $CO_2$ fluxes at the study site. We followed an iterative process in which we assessed how well the measured data fit model simulations by parameter optimization. We optimized the major parameters that govern photosynthesis, respiration, tissue turnover and turnover of soil organic matter to obtain a close match between the observed and predicted values for total biomass, grain yield, SOC, and $CO_2$ fluxes. These parameters include, but not limited to, the maximum rate of carboxylation ($V_{max25}$); the Michaelis-Menten constants for $CO_2$ and $O_2$ ($K_c$, $K_o$); the nitrogen uptake speed ($N_{up, max}$); the maximum turnover rate for each soil carbon pool ($kmax_{pool}$), the mixing efficiency of tillage ($f_{mix}$), etc. The DLEM-Ag has been intensively calibrated for different natural functional types and crop types across regions (e.g., Tian et al., 2010; Ren et al., 2011, 2012, 2016; Zhang et al., 2018). Here, we first used the default parameters to run the model and refined the parameter values to get a better match between simulated and measured results for grain yield, SOC, and $CO_2$ fluxes. The model performance was estimated following quantitative methods (Janssen and Heuberger, 1995), including the root mean square error (RMSE, Eq. 3.22), modeling efficiency (EF, Eq. 3.23), the coefficient of determination ($R^2$), and linear regression. The RMSE is a measure of the mean error between model simulation and observation.

$$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{n}(S_i - O_i)^2}{n}} \tag{3.22}$$

where $O_i$ and $S_i$ denote observed and simulated values, respectively, and n is the number of measurements. Modeling efficiency (EF) suggests how efficiently the model reproduces observations relative to the mean of observation (Karhu et al., 2012). The EF values can be positive or negative, with a maximum value of 1. The closer the value is to 1, the better the fit between simulated results and observations. If EF is negative, the model-predicted values do not capture the dynamics in time.

$$\mathrm{EF} = \frac{\sum_{i=1}^{n}(O_i - \bar{O})^2 - \sum_{i=1}^{n}(S_i - O_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \tag{3.23}$$

### 3.3.5 Model Simulation Experiment Design

The model was run at a daily time step when simulating crop development and growth. The model simulation began with an equilibration run, using 30-year (1970-1999) mean climate datasets, to get close to the initial states of carbon, nitrogen, and water pools in 1970. The equilibrium state was defined such that the year-to-year changes in carbon, nitrogen, and water pools at the site would be less than 0.1 g C $m^{-2}$, 0.1 g N $m^{-2}$, and 0.1 mm $H_2O$, respectively. In this procedure, we used the available SOC data as a benchmark and ran the model to get other initial datasets that were not measured. After the equilibrium run, we executed a 10-year spin-up run using climate data randomly selected between 1970 and 1999 to remove sudden changes caused by the shift from equilibrium to transient mode.

We then designed sixteen experiments to separate the contributions of multiple influencing factors to SOC in the long-term NT-CC system (Table 3.2). To examine the model fluctuation resulting from internal system dynamics, we first performed two "Reference" simulations driven by all factors remaining constant at 1970 levels through the 1970-2018 period under CT-CC (S1) and NT-CC (S2) treatments. The "All" simulation experiments S3 and S4 aimed to fit the historical observations for the two treatments, respectively. To attribute the relative contributions of each driving factor (i.e., precipitation, temperature, $CO_2$, and nitrogen deposition) to annual variations of SOC, we then designed four factorial simulation experiments (S5, S6, S7, S8). In each factorial experiment, this single factor was allowed to change over time, while other factors were kept constant at the level of 1970, to determine the relative importance of the climate (precipitation and temperature), $CO_2$, and nitrogen deposition. We used a simulated attributed analysis approach (Ren et al., 2016) to calculate the relative contributions of these factors. The overall change in SOC, $\Delta$SOC, was defined as the difference between the "All" simulations and the "Reference" baseline simulations (i.e., S1 vs. S3, and S2 vs. S4). The change due to each factor was the difference between the factor-specific experiments (S5, S6, S7, S8) and the baseline (S2). In addition, we defined the single factor of tillage as the difference between S1 and S2. For predictions of the future SOC trajectories, we further designed eight scenario experiments, with experiments S9, S10, S11, and S12 for four different treatments (i.e., CT-CC, CT, NT-CC, and NT) under the

RCP 2.6 scenario, and experiments S13, S14, S15, and S16 for the same four treatments under the RCP 8.5 scenario, respectively.

Table 3.2 Simulation experiments design in this study

| Simulations | Abbr. | Treatment* | Precp. | Temp. | $CO_2$ | N-Dep. |
|---|---|---|---|---|---|---|
| Reference_a | S1 | CT-CC | 1970 | 1970 | 1970 | 1970 |
| Reference_b | S2 | NT-CC | 1970 | 1970 | 1970 | 1970 |
| All_a | S3 | CT-CC | Varying1# | Varying1 | Varying1 | Varying1 |
| All_b | S4 | NT-CC | Varying1 | Varying1 | Varying1 | Varying1 |
| Precp-only | S5 | NT-CC | Varying1 | 1970 | 1970 | 1970 |
| Temp-only | S6 | NT-CC | 1970 | Varying1 | 1970 | 1970 |
| $CO_2$-only | S7 | NT-CC | 1970 | 1970 | Varying1 | 1970 |
| Ndep-only | S8 | NT-CC | 1970 | 1970 | 1970 | Varying1 |
| RCP2.6_a | S9 | CT-CC | Varying2Δ | Varying2 | Varying2 | Varying2 |
| RCP2.6_b | S10 | CT | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP2.6_c | S11 | NT-CC | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP2.6_d | S12 | NT | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP8.5_a | S13 | CT-CC | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP8.5_b | S14 | CT | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP8.5_c | S15 | NT-CC | Varying2 | Varying2 | Varying2 | Varying2 |
| RCP8.5_d | S16 | NT | Varying2 | Varying2 | Varying2 | Varying2 |

* Treatment including conventional tillage and cover crop (CT-CC), no-tillage and cover crop (NT-CC), conventional tillage (CT), and no-tillage only (NT). The CT and NT treatments were only applied to the future period (2019-2099) in experiment S10, S12, S14, and S16. In the historical period (1970-2018), these four experiments all were cover cropped. In other words, for example, the S9 and S10 were only different during the future, 2019-2099, period.

# Varying1 means historical time period, 1970-2018.

Δ Varying2 means full time period, 1970-2099.

## 3.4 Results

### 3.4.1 Model Evaluation against Field Observations

We evaluated the DLEM-Ag model by considering three variables: crop yield, SOC, and $CO_2$ fluxes. There were 50 pairs of simulated and observed crop yield values for annual tillage treatments, which spanned the historical 1970 to 2018 time period (Figure 4.3a). The regression analysis exhibited an $R^2$ of 0.86, a slope of 0.913, and an RSME of 1.1 Mg ha$^{-1}$ ($P < 0.001$), with a modeling EF of 0.79. The model yield data were reasonably well fit the measured yield data. The total SOC simulation was validated against eight observed values from 1975 to 2018 (Figure 3.3b). The regression analysis had an $R^2$ of 0.58, a slope of 0.995, and an RSME of 3.0 Mg C ha$^{-1}$ ($P = 0.027$). Considering that changes in SOC can occur slowly, the purpose of this validation was to determine if the simulated trend was in line with the observed one. The individual fitting of simulated and actual $CO_2$ flux data for each tillage system is shown in Figure 3.3c. Simulated $CO_2$ flux values were mostly within the SDs for the observed 2018 growing season $CO_2$ fluxes.

Figure 3.3d shows the results of the historical SOC model outcomes against the considerable scatter of historical SOC data points for the site. The simulated SOC values show a statistically significant correlation with observed data. Our results indicate that the DLEM-Ag model is able to capture the overall SOC trend. It should be noted that the measurements of ~13 Mg C ha$^{-1}$ (NT) and 15 Ma C ha$^{-1}$ (CT) gain and subsequent loss within a few years are likely not a realistic numerical depiction of soil carbon change in this soil.

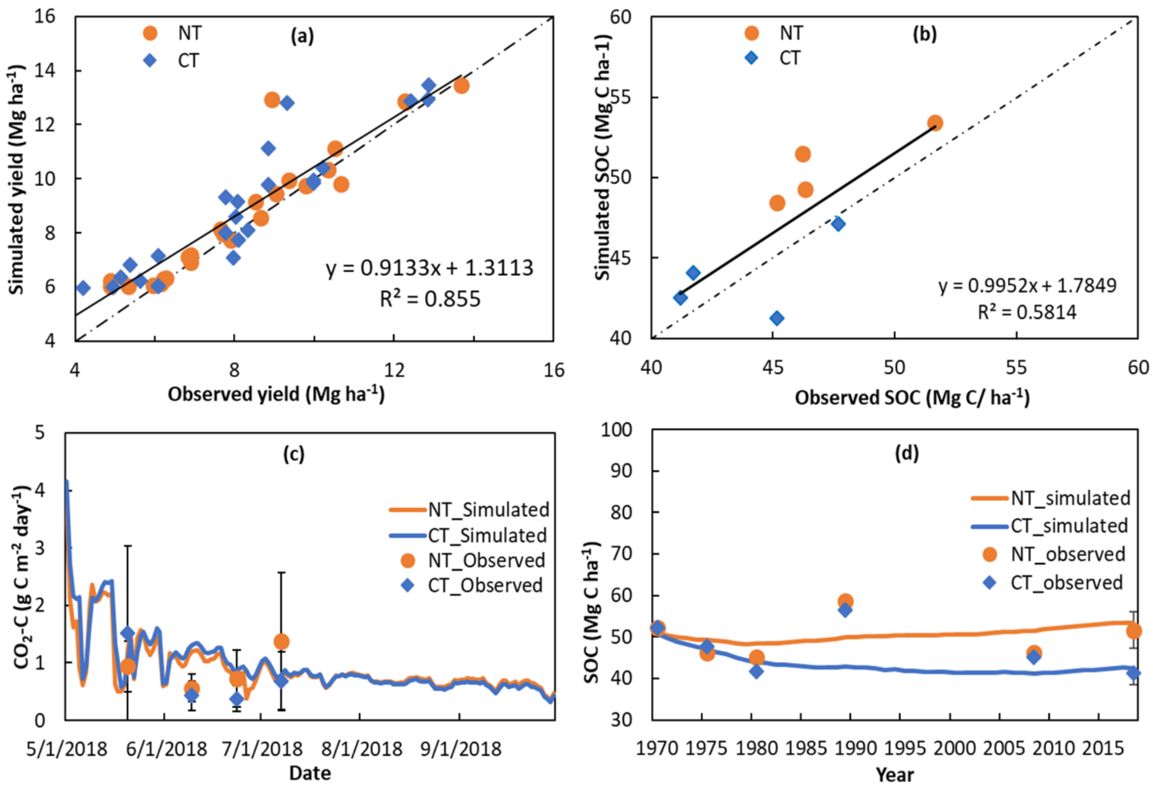


Figure 3.3 Comparison of simulated and observed (a) crop yield and (b) SOC at the 0-30cm soil depth for NT and CT systems. The dotted line represents equal values for simulated and observed data. The solid line is the linear fit of simulated versus observed data. Comparison of simulated and observed (c) $CO_2$ flux in the 2018 maize growing season and (d) SOC (0-30 cm) with time during the historical period

### 3.4.2 Soil Organic Carbon Balance under No-till and Cover Crop Management

During 1970-2018, the observed SOC showed a slightly increasing trend under NT-CC treatment and a decreasing trend under CT-CC treatment (Figure 3.3d). Our S3 and S4 simulation results caught these trends. Overall, the difference in the simulated rate of SOC change between NT-CC and CT-CC was about 0.22 Mg C ha$^{-1}$ yr$^{-1}$ during this historical period, which was similar to measured observations (Table 3.3 and Figure 3.3d). The simulation further showed that such differences between the two tillage systems diminished with time (Table 3.3). By analyzing carbon pathways in the model, we found that maize residues accounted for approximately over 65% of carbon entering the soil, followed by litterfall and cover crop biomass (Table 3.3, Figure 3.4a). The average carbon input from each pool exhibited an increasing trend from the 1970s to the 2010s, indicating growing productivity in the study. Moreover, our simulation also caught the significant decline of SOC in both systems and the low carbon input from cover crops during the 1970s. The simulated total carbon inputs were similar in NT-CC and CT-CC systems. According to the simulated pathway analysis, most of the incoming carbon was lost through $CO_2$ emissions, with higher rates in the CT-CC system than in the NT-CC system (Figure 3.4a). This discrepancy led to different carbon sequestration rates between the CT-CC and NT-CC systems (Figure 3.4b), although carbon loss through leaching was higher with NT-CC compared with CT-CC (Figure 3.4b). It should be noted that the magnitude of leached carbon was higher than that of the net SOC change rates, suggesting the potential to enhance SOC sequestration by better soil water management. However, the leached carbon showed an increasing trend over the past several decades, corresponding to the increased precipitation at the site. In addition, the tillage-only experiment showed that tillage decreased SOC by about 0.18 Mg C $ha^{-1}$ $yr^{-1}$ compared to the NT due to greater soil $CO_2$ emission (Figure 3.5a).

Table 3.3 Simulated results of decade-averaged major carbon input (i.e., litterfall (LF-C), cover crop (CC-C), and maize residues (CR-C)), output (i.e., $CO_2$ ($CO_2$-C), leaching organic carbon (L-C) pools), and net SOC changes

| NT-CC | C input | | | C output | | | C sequestered |
|---|---|---|---|---|---|---|---|
| (Mg C $ha^{-1}$ $yr^{-1}$) | LF-C | CC-C | CR-C | $CO_2$-C | L-C | Others | ΔSOC |
| 1970s | 0.81 | 0.24 | 2.64 | 3.56 | 0.18 | 0.22 | -0.27 |
| 1980s | 0.81 | 0.64 | 3.31 | 4.33 | 0.10 | 0.23 | 0.11 |
| 1990s | 0.84 | 0.92 | 3.21 | 4.47 | 0.22 | 0.22 | 0.07 |
| 2000s | 0.87 | 1.01 | 3.36 | 4.71 | 0.16 | 0.24 | 0.13 |
| 2010s | 0.93 | 1.30 | 3.51 | 5.06 | 0.23 | 0.27 | 0.18 |
| 1970~2018 | 0.85 | 0.82 | 3.21 | 4.43 | 0.18 | 0.24 | 0.05 |
| CT-CC | C input | | | C output | | | C sequestered |
| (Mg C $ha^{-1}$ $yr^{-1}$) | LF-C | CC-C | CR-C | $CO_2$-C | L-C | Others | ΔSOC |
| 1970s | 0.81 | 0.24 | 2.65 | 3.98 | 0.15 | 0.21 | -0.65 |
| 1980s | 0.81 | 0.62 | 3.28 | 4.63 | 0.06 | 0.21 | -0.19 |
| 1990s | 0.84 | 0.89 | 3.21 | 4.62 | 0.16 | 0.21 | -0.06 |
| 2000s | 0.86 | 0.96 | 3.34 | 4.86 | 0.12 | 0.21 | -0.03 |
| 2010s | 0.93 | 1.28 | 3.51 | 5.21 | 0.16 | 0.26 | 0.09 |
| 1970~2018 | 0.85 | 0.80 | 3.20 | 4.66 | 0.13 | 0.22 | -0.16 |

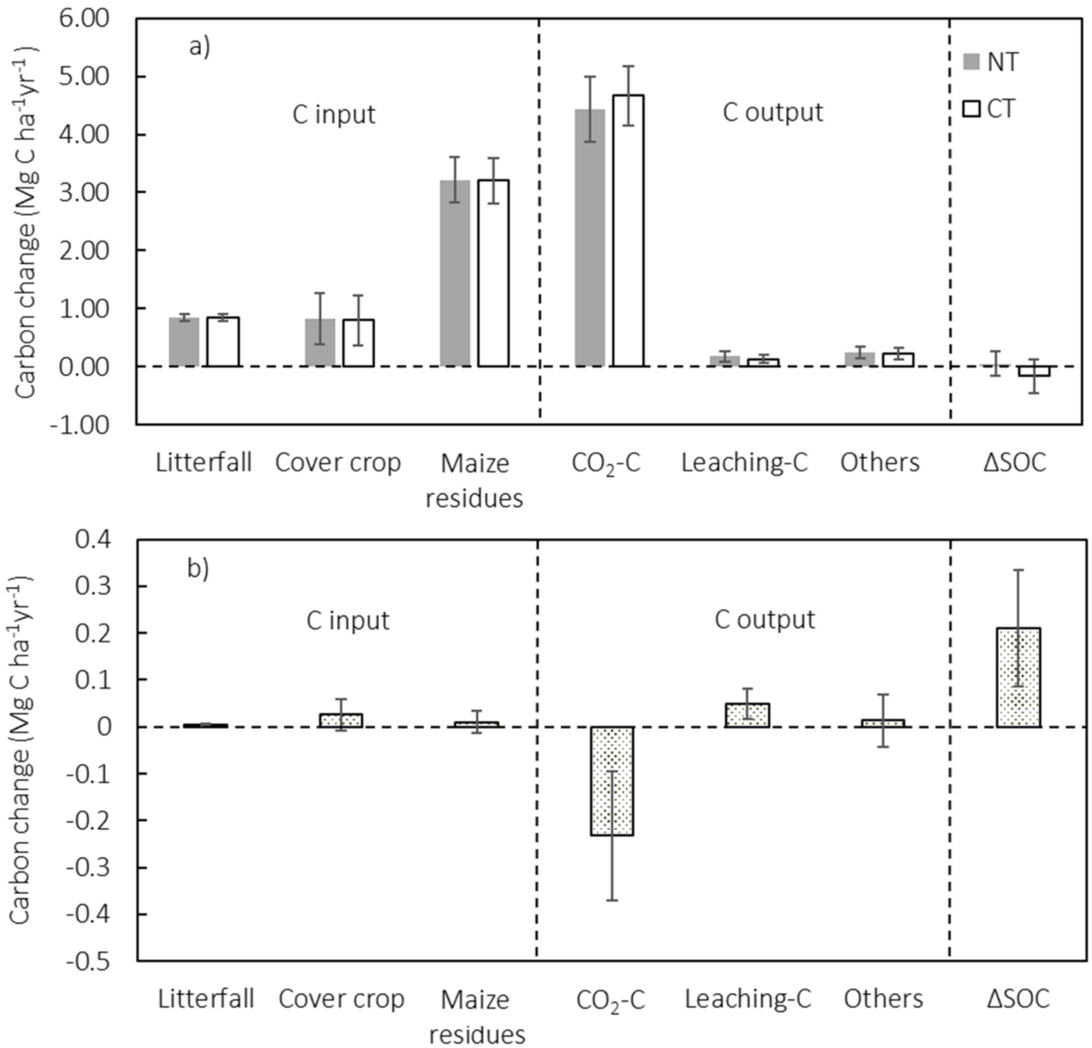


Figure 3.4 (a) Simulated results of 49-year averaged change in each carbon pool with NT and CT production systems, and (b) the associated differences between NT and CT for each carbon pool

### 3.4.3 Relative Contribution of Environmental Factors to Long-term Variations in SOC During the 1970-2018 Time Period

The single-factor simulation experiments enabled an examination of SOC changes induced by temperature, precipitation, $CO_2$, and nitrogen deposition (Figure 3.5a). Our results indicated that the relative contributions of the four environmental factors varied greatly over the historical period. Atmospheric $CO_2$ enhanced SOC by 0.036 Mg C $ha^{-1}$ $yr^{-1}$ in NT-CC over the 49-year period, as compared to the baseline (Figure 3.5a). Annual temperature and precipitation in the study area both exhibited an increasing trend over the past several decades (Figure 3.1a, c). However, the temperature-only and precipitation-only experiments showed contrasting SOC responses (Figure 3.5a). Compared with precipitation, changes in temperature better stimulated winter cover crop growth, resulting in more carbon entering the soil (Figure 3.5b). This suggested that temperature was more pronounced than precipitation as a factor limiting winter cover crop growth at this site. Increased temperature ultimately resulted in positive effects of warming on SOC, although the warming trend also increased $CO_2$ emissions (Figure 3.5b). Increased precipitation led to negative effects on SOC. Considering the treatments are crop nitrogen nutrition sufficient, the negligible impacts of nitrogen deposition on SOC were no surprise (Figure 3.5).

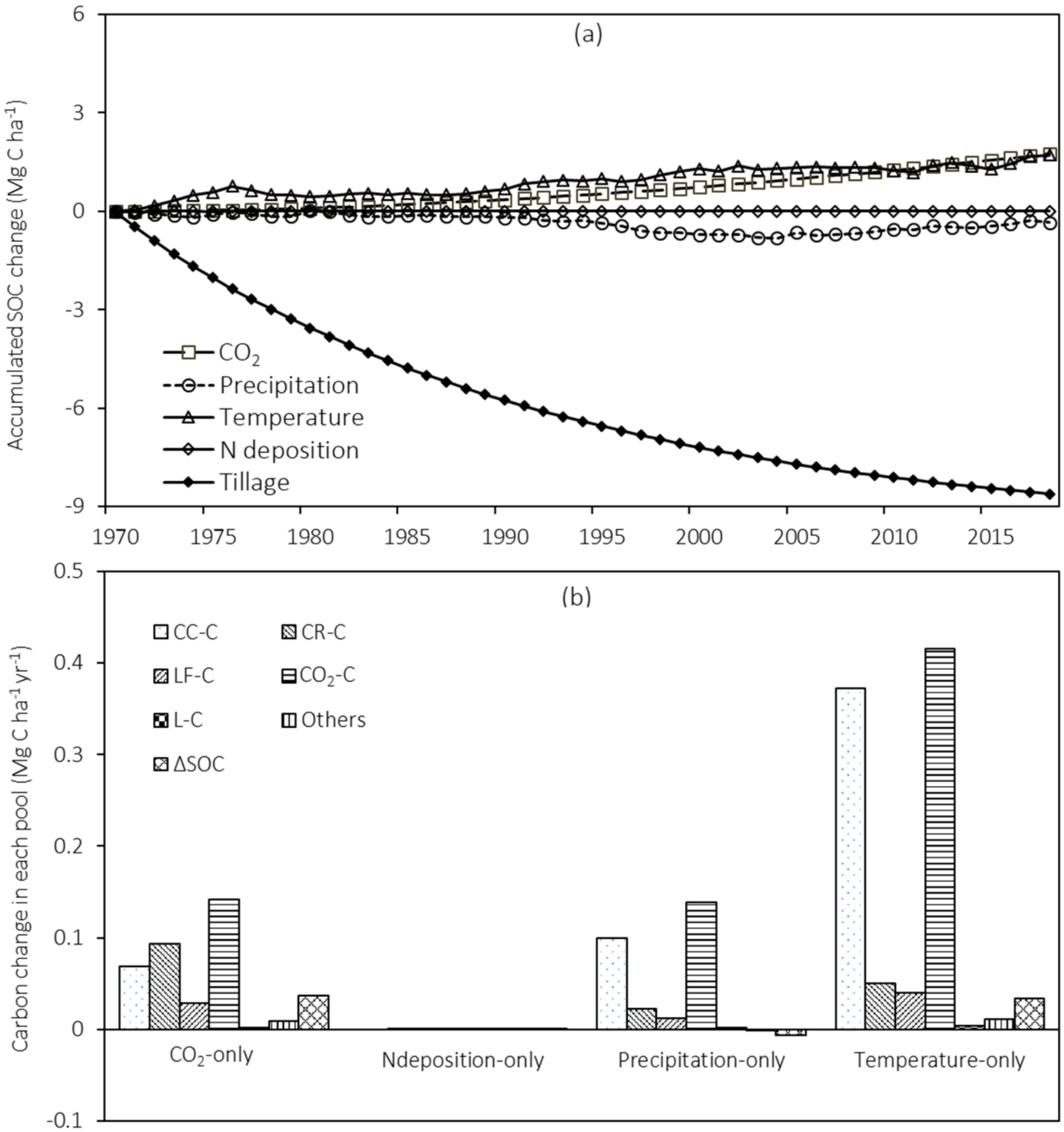


Figure 3.5 (a) The accumulated SOC change from 1970 to 2018 driven by single factor simulation; (b) Annual carbon change in each carbon pool for each single factor simulation as compared to the baseline simulation (CC-C, CR-C, LF-C, $CO_2$-C, L-C represent cover crop, maize residues, litterfall, $CO_2$, and leached carbon pools, respectively)

### 3.4.4 Potential Trajectories of SOC under Future Climate Scenarios

For the future, predicted SOC at the 0-30 cm depth exhibited significant differences due to tillage ($p<0.0001$) under the RCP 2.6 scenarios (Figure 3.6a), losing 0.017 ± 0.014 Mg C $ha^{-1}$ $yr^{-1}$ and 0.002 ± 0.016 Mg C $ha^{-1}$ $yr^{-1}$ in the CT-CC and NT-CC treatments, respectively. This represents a 3% decline in SOC in the CT-CC treatment and 0.3% in the NT-CC treatment over the 2019-2099 period. In addition, 17 out of 36 simulations under the GCM-generated RCP 2.6 scenarios predicted an increasing trend in SOC with the NT-CC treatment. In comparison, only four simulations predicted a slight growing trend in SOC with the CT-CC treatment. Under the GCM-generated RCP 8.5 scenarios, all simulations predicted an increase in SOC at the 0-30 cm depth in both treatments, although the increases are not uniform (Figure 3.6b). On average, the predictions showed that SOC would increase at a rate of 0.089 ± 0.021 Mg C $ha^{-1}$ $yr^{-1}$ with the NT-CC treatment, significantly higher than that with the CT-CC treatment (0.058 ± 0.017 Mg C $ha^{-1}$ $yr^{-1}$). We found the main reason leading to SOC increases under RCP 8.5 scenarios could be enhanced winter cover crop growth, although carbon loss through decomposition showed an increasing trend (Figure S3.3). During the simulated period, the predicted incoming carbon from cover crop biomass increased in both CT-CC and NT-CC plots (0.026 ± 0.0034 and 0.023 ± 0.0032 Mg C $ha^{-1}$ $yr^{-1}$, respectively, Figure S3.1). The incoming carbon from maize residues also increased but at relatively lower rates (0.007 ± 0.0006 and 0.006 ± 0.0039 Mg C $ha^{-1}$ $yr^{-1}$ with CT-CC and NT-CC treatments, respectively, Figure S3.2). We further conducted sensitivity experiments by excluding winter cover crops from the system and found that SOC declined in all treatments and climate scenarios (Figure 3.7). This indicates that if cover crops were not applied, the higher temperature would induce greater SOC loss regardless of tillage, and the combination of cover crops and NT can be effective management for climate change mitigation in terms of enhancing SOC sequestration.

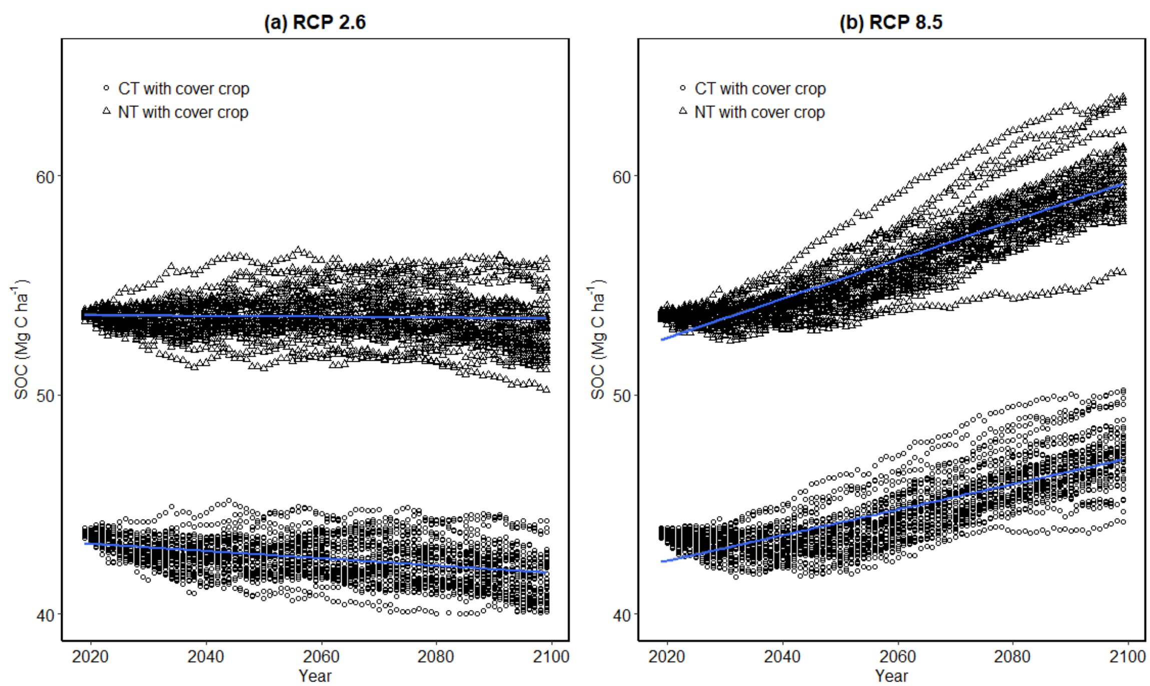


Figure 3.6 Predicted SOC changes during 2019-2099 at the 0-30 cm depth for NT and CT (with cover crop) under (a) RCP2.6 and (b) RCP8.5 scenarios. Blue lines are average trends

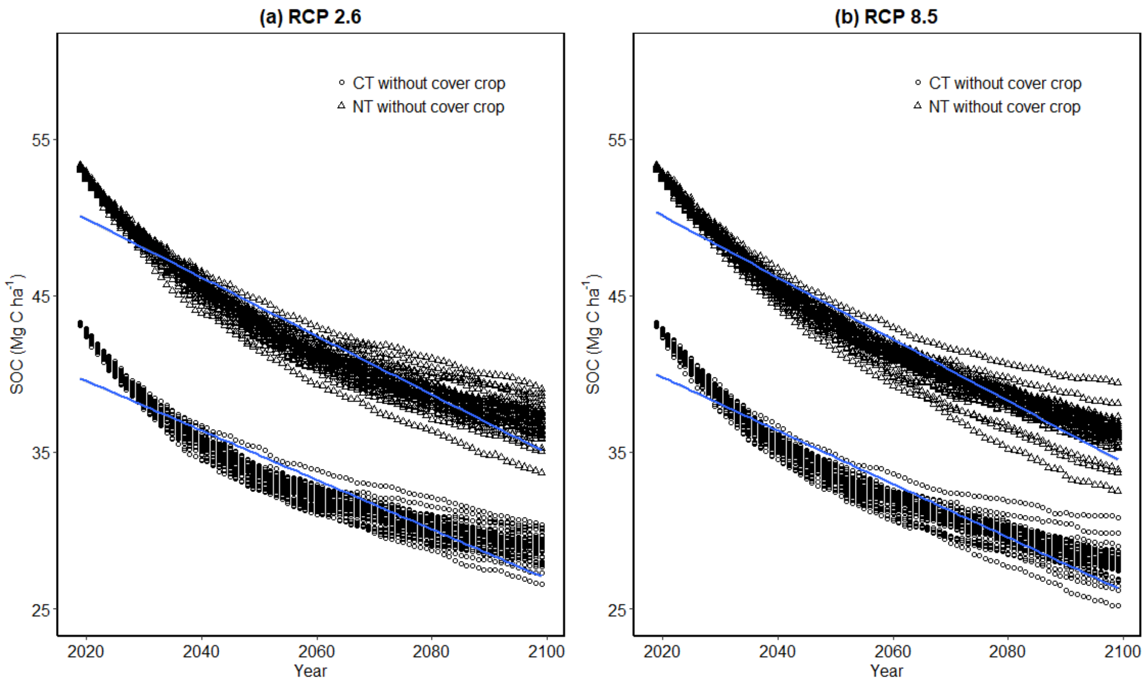


Figure 3.7 Predicted SOC changes during 2019-2099 at the 0-30 cm depth for NT and CT (without cover crop) under (a) RCP2.6 and (b) RCP8.5 scenarios. Blue lines are average trends

## 3.5 Discussion

### 3.5.1 Synergistic Effects of No-tillage and Cover Crops on SOC

Generally, our simulated changes in SOC caught the observed overall trend at this site, although they did not reflect the high variability in observations over the years. In fact, changes in SOC can occur slowly (Maas et al., 2017). In such a long-term study that includes various people collecting and analyzing samples, many factors could possibly lead to the variability in measurements. For example, how the surface residue was scraped off can affect SOC determination (Maas et al., 2017). Although all samples went to a common depth of 30 cm, the samples were divided into different depth increments in different years. Using different SOC analysis laboratory/methods and disregarding spatial variability during the sampling scheme can also lead to errors in reporting SOC (Olson et al., 2014). In addition, the stratification of SOC can add variability to the results.

Changes in soil carbon stocks are determined by the balance between carbon entering the soil via plant detritus and carbon losses through microbial decomposition, leaching, and erosion. The model simulated annual change in SOC was highly correlated to annual carbon input ($r$ = 0.84, $p < 0.0001$, data not presented). This result parallels studies that report a positive correlation between the amounts of carbon input and SOC stock (Kong et al., 2005; Nash et al., 2018). Averaged over 49-year simulations, the incoming carbon from maize residues showed little differences due to tillage treatment yield differences, with the NT soil receiving slightly more carbon than the CT plots (Table 3.3). This reflects better plant growth in the NT system, which corresponds with earlier results published for the study area (Grove et al., 2009). No-till soils usually contain a greater amount of soil water than tilled soils due to less evaporation and higher infiltration (Phillips, 1984), which can carry crops through periods of short-term drought without detrimental stress (Blevins et al., 1971). The simulated cover crops also had greater biomass production in the NT system than in the CT system (Figure 3.5b). The cause of this difference in cover crop biomass production due to tillage is unclear. One possible explanation could be that the higher soil moisture content with NT, as compared to CT, also benefited winter cover crop growth. In addition, greater SOC build-up in NT could provide greater mineralizable N. This may be important for cover crop growth because cover crops are not fertilized - they just receive some residual fertilizer N and N mineralized from SOM. Apart from tillage, other management practices, including nitrogen fertilizer rate, were identical in the two tillage system treatments.

Considering the evidence in the literature showing that NT usually results in greater SOC sequestration than for tilled soils (Franzluebbers, 2010; Bai et al., 2019), the corroborative evidence in this study is no surprise. However, the early years of the experiment witnessed a decline in SOC in both systems, with a greater loss with the CT than the NT (Figure 3.3b, Table 3.3). One possible reason for the carbon loss could be due to the conversion from perennial grass to annual crop production. Moreover, there were very poor stands of cover crops in the 1970s because of hand broadcast seeding (Blevins et al., 1977), which would have resulted in low carbon input from cover crops. The simulated results caught this declining trend of SOC during the early years with lower cover crop biomass inputs (Figure 3.3b, Table 3.3). We further found that when the cover crops were excluded from the system in the simulations under future scenarios, the projected SOC would significantly decrease (Figure 3.7). This is consistent with findings that crop rotation management has a greater impact on SOC than tillage management (Nash et al., 2018) because cover crops increase crop rotation diversity and provide additional carbon. However, what applies to this study site should be cautiously interpreted, as the effectiveness in promoting productivity and carbon sequestration through NT and cover crops is spatiotemporally heterogeneous (Liu et al., 2014; Pittelkow et al., 2015; Poeplau and Don, 2015; Paustian et al., 2016; Bai et al., 2019).

Our simulation results showed that $CO_2$ emission was the main pathway accounting for carbon loss from the soil. Compared with NT, tillage causes incorporation of crop residues in the soil, breakdown of soil aggregates, and exposure of protected SOC, which all render the organic matter more accessible to decomposers (Fiedler et al., 2016) and results in higher $CO_2$ emission. The amount of cover crop biomass is, to some extent, a reflection of its growth stage. Cover crops at termination in the CT system will be at an earlier, perhaps more succulent growth stage (i.e., lower C:N) than that of cover crops at termination in the NT system. This would result in a faster rate of cover crop residue decomposition in the CT system (Munawar et al., 1990). The lower $CO_2$ emission from the NT treatment largely accounts for the lower soil carbon loss compared with the CT treatment, although leaching of organic carbon is greater in NT than in CT (Figure 3.4b).

Greater leached carbon loss in NT can be ascribed to the frequent occurrence of macropores (Kleinman et al., 2009) and a general improvement in soil infiltration capacity (So et al., 2009). Further, there is better surface water interception under NT in such well-drained and moderately permeable soils, which would drive additional water infiltration. This is supported by studies showing NT soil had higher saturated hydraulic conductivity than tilled soil (Blevins et al., 1983). However, NT impacts on soil hydraulic properties can be highly site-specific (Blanco-Canqui et al., 2017; Schwartz et al., 2019; Stone and Schlegel, 2010); depending on soil texture, the measurement time, the duration of NT implementation, cropping system, and interactions with other management practices. In a recent meta-analysis, Daryanto et al. (2017) reported that NT soil management tended to increase leachate nitrate load, relative to tillage, but with similar leachate nitrate concentration, suggesting greater soil water infiltration flux in NT.

### 3.5.2 Climate Change ($CO_2$, temperature, and precipitation) Effects on SOC

*$CO_2$ effects*: Climate change could alter soil carbon storage because changes in atmospheric $CO_2$ concentration, temperature, and precipitation will affect plant biomass carbon inputs and soil carbon decomposition rates. The single factor simulations suggested a slight positive effect of increasing $CO_2$ on SOC over the past 49 years, consistent with

previous model results (e.g., Ren et al., 2012; Banger et al., 2015; Tian et al., 2011, 2012, 2015). Elevated $CO_2$ ($eCO_2$) benefited plant biomass production in the NT-CC treatment, and subsequent carbon input to the soil as the substrate for decomposition (Figure 3.5b), which confirmed the idea that conservation management in the $eCO_2$ environment increases soil carbon storage by increasing cumulative residue input (Prior et al., 2005). There is much evidence that $eCO_2$ stimulates plant carbon accumulation (Kimball, 1983; Luo et al., 2006; Jones et al., 2014; Wijewardana et al., 2016), with more consistent positive effects with C3, as opposed to C4, plants (Kimball, 2016; Leakey et al., 2009). Maize, as a C4 plant, is indirectly stimulated by $eCO_2$ and usually occurs in situations of drought (Leakey et al., 2009). One possible explanation for the stimulation of maize residue under the $CO_2$-only experiment could be that the maize growing season in 1970 was relatively drier than the average (Figure 3.1c), and 1970 was used as the reference weather year.

Another possible mechanism might be that the stimulation of rye cover crop due to $eCO_2$ synergistically benefited maize growth because the cover crop provided more carbon input to soils and a greater supply of easily metabolized substrates. This may stimulate the decomposition of native SOC due to the priming effect (Blagodatskaya and Kuzyakov, 2008), giving faster decomposition rates (Van Groenigen et al., 2014). A recent review by Kuzyakov et al. (2018) suggested that an annual increase in plant productivity by 13-20% $yr^{-1}$ with $eCO_2$ would give total SOC increases of only 1.2 to 2.2%. However, some studies show that $eCO_2$ would shift the quality of plant shoot and root residue to higher C/N ratios (Norby et al., 2001; Luo et al., 2006; Feng et al., 2015), thereby lowering decomposition rates (Marhan et al., 2008). As atmospheric $CO_2$ concentration elevated from 325 ppm to 408 ppm in the $CO_2$-only simulation, the incoming carbon from maize and cover crop residue biomass increased by 0.35 Mg C $ha^{-1}$ and the C/N ratio of litter increased by 5.1% while $CO_2$ emission increased by 0.32 Mg C $ha^{-1}$.

In addition, the priming effect, i.e., enhanced mineralization of nitrogen from native SOM, can be more pronounced when soil inorganic nitrogen content is limited but be alleviated if it is sufficient (Kuzyakov et al., 2018). Generally, $eCO_2$ can promote SOC accumulation in the presence of nitrogen fertilizer application rates above the typical rates from atmospheric nitrogen inputs (Van Groenigen et al., 2006). The sufficient nitrogen rate in this study would retard the priming effect on SOM decomposition. Crop residues in the NT treatment were left on the soil surface, minimizing contact with soil microbes, and often resulting in lower residue decomposition rates compared with those for residues in tilled soils. Therefore, it could be expected that the retardation of the priming effect in the NT treatment would otherwise have a positive effect on SOC accumulation.

*Temperature effects*: The warming effects on SOC stocks have been reported to be various, with positive, negative, and neutral impacts across observations (Crowther et al., 2016). The elevated temperature would enhance soil heterotrophic respiration, hence increased soil carbon loss (Black et al., 2017), and would also stimulate plant growth and greater subsequent soil carbon input (Cowles et al., 2016). Warming generally enhances carbon fluxes to and from the soil (Lu et al., 2013). Crowther et al. (2016) found that warming effects on organic carbon in the top 10 cm soil depend on the size of the initial carbon stock, with a threshold of 20-50 Mg C $ha^{-1}$ below which minor losses due to accelerated decomposition may be offset by concurrent increases in soil carbon from enhanced plant growth. Additionally, temperature affects evapotranspiration and subsequent soil water content, which indirectly influences decomposition rates. A meta-analysis showed that warming-induced soil moisture deficits could partly offset the positive impacts of warming on soil respiration (Wang et al., 2014).

Our results from the temperature-only simulation showed a positive SOC response to elevated temperature, which can largely be attributed to the significant increase in winter cover crop growth due to warming (Figure 3.5b). Ruis et al. (2019) found that winter cover crop biomass production increased as temperature increased in a humid region, winter rye commonly produced 5.42 Mg ha$^{-1}$ in the warm zone as compared to 2.9 Mg ha$^{-1}$ in the cold zone. They suggested that tillage did not generally affect cover crop biomass production. This would support our findings that rye biomass production increased under the high emission scenarios (RCP 8.5). Considering carbon input is positively correlated with SOC content (Nash et al., 2018), there would be a high probability that the changes in SOC are sensitive to cover crop production. Basche et al. (2016) also modeled growth in a winter rye cover crop in Iowa under RCP 4.5 scenarios. The simulated SOC stock decreased in the NT-CC system but at a significantly lower rate than that in the NT without cover crops. One reason could be an inadequate supply of biomass (~1.3 Mg ha$^{-1}$ yr$^{-1}$) due to the relative dryer and cooler weather in their study area. Additionally, their simulations did not consider the effect of e$CO_2$ on cash crops and cover crop growth, which has the potential to offset the effects of future climate change to some degree.

However, we acknowledge that our model simulation might overestimate the effect of cover crop on SOC to some extent. For example, the DLEM-Ag considers a linear negative effect of $CO_2$ on stomatal conductance, according to Ainsworth and Long (2005). While the knowledge of how elevated $CO_2$ will affect photosynthesis is still under debate, the linear model might overestimate the $CO_2$-fertilization effects. In addition, uncertainties exist in the model because we considered cover crops as a rotational species that leaves all its biomass in the field at termination. We did not account for the annual growth of weeds, which was difficult to be quantified due to the lack of observations. When the cover crops were removed from the system in the future sensitive scenarios, there was only carbon output but no input during the fall-winter time period after maize harvesting and until the next maize planting. This would also cause an overestimate of SOC loss in simulations without cover crops.

*Precipitation effects*: Moisture facilitates accelerated decomposition rates and stimulates plant productivity. Because of the complex effects of soil moisture on the production and decomposition of plant biomass, its influence on SOC stocks is still unclear (Falloon et al., 2011). A meta-analysis of precipitation manipulation experiments showed that increased precipitation stimulated soil respiration and plant biomass by an average of 45% and 12%, respectively (Wu et al., 2011). Our results from the precipitation-only simulation showed that the increased decomposition with precipitation exceeds that of the plant biomass carbon inputs (Figure 3.5b), probably because precipitation is less limiting to plant growth at the study site. The timing and frequency of precipitation can also have large effects on plant growth and decomposition (Knapp et al., 2008). The GCMs projects that much of the precipitation increase will occur in heavier events in the southeastern U.S. (Melillo et al., 2014), suggesting more frequent flood and drought events. Excessive rainfall and excessive drought could both reduce crop productivity (Li et al., 2019) and subsequently reduce biomass carbon inputs. Therefore, the negative impact of precipitation on SOC could be amplified by changes in future precipitation regimes. Our simulations under future scenarios, however, did not consider the effect of these extreme events on crop growth, which might offset some of the negative impacts.

### 3.5.3 Uncertainties

This study quantified the long-term synergistic effects of NT and cover crops in building the capacity of agricultural soils to sequester carbon. However, due to the scarcity of observational SOC data from deeper soil profile at this site, we were unable to calibrate the model and evaluate performance at deeper soil depths. This hinders us from understanding whether their synergetic benefit on SOC is limited to the surface layer at this site. Although Grove et al. (2009) reported SOC stocks to 1 m, with NT exhibited higher than CT in 2008, this issue is still under debate (Bai et al., 2019; Powlson et al., 2014) and needs to be addressed with more consistent measurements deeper in soil profiles in future field research.

There are also several limitations in this study that may bring uncertainties to our SOC change estimates. Some ecosystem processes are not well represented in the current model due to a lack of observations/measurements and associated knowledge of mechanisms. For example, root-derived carbon (root biomass C plus rhizodeposition C) significantly contributes to SOC stocks (Johnson et al., 2006). However, it is still challenging to accurately estimate the amount of carbon allocated belowground as this relies on systematic measurements of root biomass. Additionally, there is a lack of information regarding the direct effects of $eCO_2$ on soil carbon turnover rate, i.e., the priming effect (Van Groenigen et al., 2014), and DLEM-Ag might underestimate decomposition. Besides, $eCO_2$ would significantly promote mycorrhizal growth (Treseder, 2004), which potentially enhances soil aggregation and thereby protects SOM from microbial decomposition (Rillig, 2004). The increase in mycorrhizal growth with $eCO_2$ will also co-metabolically accelerate SOM decomposition for nitrogen assimilation (Lindahl and Tunlid, 2015). The model uses three microbial carbon pools to represent microbial growth but has not included the description of mycorrhizal-associated processes. Future experiments and observations on soil microbial community development are urgently needed to narrow this knowledge gap.

Our future prediction did not account for changes in planting dates and cultivars for either maize or winter rye cover crops, which are among the essential factors affecting crop production (Sacks and Kucharik, 2011). A warming climate might shift maize planting to earlier dates (Sacks and Kucharik, 2011), implying earlier termination of winter cover crops and less biomass accumulation. In this case, our results might overestimate cover crop production under RCP8.5 scenarios. Furthermore, given more frequent and severe climate extreme events predicted for the future, model improvement with a rational representation of extreme climate effects on crop growth and soil could further improve model simulations of carbon dynamics in response to different management practices. In addition, higher $N_2O$ emissions can occur with NT or cover crops (Huang et al., 2018; Basche et al., 2014), which may partially offset positive effects on SOC balances. Future studies are needed to further emphasize these concerns.

## 3.6 Conclusions

This study offers the first attempt to examine the synergistic effects of no-tillage and cover crops on soil carbon dynamics in a continuous maize cropping system by integrating long-term field observations, agroecosystem modeling, and future climate scenarios. Our results show that the improved model is able to simulate soil organic carbon dynamics with different tillage practices and environmental conditions in agroecosystems. Our results demonstrate that NT and cover crops work synergistically to increase SOC, mainly via slowing down soil carbon decomposition rates and increasing cumulative carbon inputs, respectively. Therefore, combining NT and cover crops could serve as a viable adaptive strategy to mitigate climate change through soil carbon sequestration in agroecosystems. Factorial analyses suggest that soil carbon sequestration is highly associated with carbon inputs from retained crop residues. Our predictions for future climate scenarios also suggest that cover crops are crucial to maintaining or increasing SOC stocks as NT alone is not enough, even in a continuous maize system with large residue inputs. Nevertheless, the extent to which NT and cover crops benefit SOC sequestration is temporally variable and depends on other management factors and site-specific environmental conditions. There is a large impact of spatial heterogeneity on the capability of NT and cover cropping to enhance crop productivity and reduce other greenhouse gas emissions (e.g., $CH_4$ and $N_2O$). Therefore, regional assessments regarding NT and cover crop effects on soil biogeochemical dynamics are essential to better understand and quantify the role of land management practices in the global effort to combat climate change.

# CHAPTER 4. MODELING CROP YIELD AND GREENHOUSE GAS EMISSIONS IN KENTUCKY CORN AND SOYBEAN CROPPING SYSTEMS DURING 1980-2018

## 4.1 Introduction

The challenge of modern farming in response to climate change and population growth is to simultaneously improve crop yield and reduce greenhouse gas (GHG) emissions. Among the portfolio of management options for climate change adaptation and mitigation, conservation management practices, such as no-tillage (NT), have been promoted to decrease environmental side effects (e.g., GHG emissions and soil degradation) while ensuring agricultural productivity in the long-term (Lal, 2013). No-tillage is a system that avoids tillage of the soil and leaves crop residues on the soil surface. It can benefit the function and quality of soil in many situations and, therefore, crop production (Blanco-Canqui and Ruis, 2018; Skaalsveen et al., 2019). However, NT effects on crop yield and GHG emissions are highly variable (Pittellow et al., 2015; Huang et al., 2018). With the growing adoption of NT worldwide, it is becoming controversial whether sustainable crop production and its management practices can effectively enhance food security and mitigate GHG emissions in the long term (Powlson et al., 2014; VandenBygaart, 2016).

No-tillage can reduce risks of soil degradation from erosion (Montgomery et al., 2007; Derpsch et al., 2010), thus holding more soil organic carbon (SOC) and water to maintain or improve soil quality compared to conventional tillage (CT) practices (Huggins and Reganold, 2008). This is especially important in Kentucky because soils in this region are vulnerable to water erosion under intensive tillage, considering higher amounts and more intense rainfall that are characteristic of the region (Triplett and Dick, 2008; USDA, 2015). The adoption of NT continues rising in recent decades. According to the USDA Agricultural Resource Management Survey, NT accounts for 68% of the total acreage of Kentucky's cropland (2017).

Many studies have reported positive yield responses to NT for rainfed crops in dry climates (Pittelkow et al., 2015; Huang et al., 2018) or in humid conditions with moderate- to well-drained soils (DeFelice et al., 2006; Triplett and Dick., 2008). However, in more mesic climates or poorly drained soils, the yield responses could be more variable. Soil water conservation and retention can be a benefit for NT management under water-limited conditions (Farooq et al., 2011). Still, the potential for soil waterlogging and delayed soil warming in spring can be detrimental to crop growth (Licht and Al-Kaisi, 2005). While studies in Kentucky generally have reported improved productivity with NT at several sites (Blevins et al., 1971; Díaz-Zorita et al., 2004; Grove et al., 2009), NT yield outcomes have not been quantified at the state level.

No-tillage can reduce soil $CO_2$ emissions primarily due to: 1) less soil disturbance that keeps SOC unexposed (Rastogi et al., 2002); 2) improved soil aggregate stability that protects SOC from microbial attack (Abdalla et al., 2013); 3) a lower soil temperature (Lu et al., 2016). However, greater soil moisture availability with NT could also enhance the microbial activity, thus increasing $CO_2$ emissions (Plaza-Bonilla et al., 2014b). A recent global meta-analysis suggested that the reduced $CO_2$ under NT could diminish with the increasing duration of NT management when the system reaches a new equilibrium at higher SOC stocks (Huang et al., 2018).

Because soil water status and bulk density in NT soils are usually higher compared to CT soils, it has been claimed that higher $N_2O$ emissions can occur with NT (Smith et al., 2001). In contrast, NT soils may have a slower N mineralization rate than CT soils, leading to less $NH_4^+$ and $NO_3^-$ contents in NT soils and, therefore, lower $N_2O$ emissions (Dick et al., 2008; Almaraz et al., 2009a). In the long-term, the adoption of NT substantially modifies soil physical properties with improved soil aggregation and aeration status, and consequently reduce $N_2O$ emissions (Six et al., 2004; van Kessel et al., 2013).

Generally, climate conditions, soil texture classes, and duration of NT management could be the major factors that affect the responses of agricultural systems to NT management (Pittellow et al., 2015; Blanco-Canqui and Ruis, 2018; Cusser et al., 2020). However, a knowledge gap still exists regarding NT effects on crop yield and GHG emissions at large spatial scales. Field experiments are an invaluable approach in revealing the impact of management practices on agronomic and environmental variables with great credibility (Plaza-Bonilla et al., 2018). However, there are limitations when interpreting and applying site-specific findings. Although meta-analysis is a useful tool to test the linear relationship between site-specific conditions and crop yield/GHG emissions, it cannot assess the spatiotemporal magnitude and pattern as affected by management practices. Process-based models provide an opportunity to overcome these limitations (Lutz et al., 2019) and to help establish management decisions (Ludwig et al., 2011). Therefore, the objective of this study was to: 1) assess the NT effects on crop yield and GHG emissions in Kentucky croplands during 1980-2018 using an agroecosystem model (DLEM-Ag); 2) examine the environmental factors (i.e., climate and soil) that regulating the NT effects.

## 4.2 Materials and Methods

### 4.2.1 Description of the Study Area

Kentucky lies in the east-central portion of the United States. The Ohio River forms a northern border and the Mississippi River a western border. Kentucky experiences a humid subtropical climate with an oceanic climate in the highlands of the southeast. Hot summers and cold winters occur typically, with a gradual increase in warmth in the southern regions. It receives a high amount of rainfall, with an average of 1143 mm of annual rainfall and an increase from the north to south. Soybean and corn are the two major row crops grown in Kentucky, accounting for about 25% of the total cropland.

### 4.2.2 Input Driving Data

#### 4.2.2.1 Climate, $CO_2$ and Nitrogen Deposition Data

The daily climate data we used to drive the model were derived from the Daymet Version 3 model output data at a resolution of 1-km × 1-km covering Kentucky from 1980 to 2018 (Thornton et al., 2016), including maximum and minimum temperature, precipitation, shortwave radiation, and vapor pressure. The historical $CO_2$ concentration dataset was retrieved from NOAA/GML (www.esrl.noaa.gov/gmd/ccgg/trends/). Gridded N deposition maps were adapted from the North American Climate Integration and Diagnostics – Nitrogen Deposition Version 1 (NACID-NDEP1) dataset (Hember 2018).

#### 4.2.2.2 Crop Distribution Map

The crop distribution map used in Kentucky was created by using the USDA-NASS Cropland Data Layer (CDL) datasets. Using the available CDL datasets for Kentucky (2008-2018), we first estimated the maximum distribution of corn and soybean between

2008 and 2018 at the 30-m layers. Then, we calculated the fractions of corn and soybean, respectively, at each 1-km pixel based on the 30-m layers (Figure 4.1). We eliminated grids with less than 5% of corn or soybean area during the model simulation and assumed that corn and soybean had the maximum cultivated area during the simulation period.

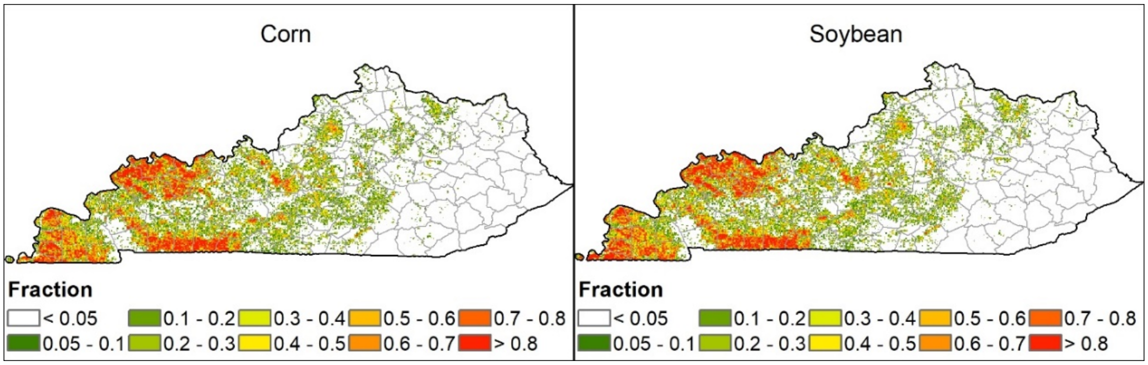


Figure 4.1 Fractional distribution of corn and soybean in Kentucky

#### 4.2.2.3 Tillage and Other Agricultural Management Practices

We obtained county-level tillage information from the Conservation Technology Information Center's (CTIC; https://www.ctic.org/) National Crop Residue Management Survey (CRM). The tabular data provides the acreages and percentages of five tillage types implemented for all crops, including corn and soybean. For simplification, we grouped the five major tillage types into three categories, i.e., no-tillage (NT), reduced tillage (RT, including ridge tillage, mulch tillage, and reduced tillage), and conventional tillage (CT). We used county acreages in combination with the CDL-derived cropland layer to estimate the spatial distribution of conventional and conservation tillage percentages for corn and soybean, assuming each pixel within a county has the same portions of the tillage-specific area. We reconstructed annual tillage maps from 1989-2011 based on the CRM dataset and made the assumptions that the tillage maps of other years are similar to the nearest year. In addition, we also generated two ideal tillage maps (all NT vs. all CT) with all the corn/soybean under one tillage regime for sensitivity analysis. Crop specific N fertilizer use data at the state level were derived from the USDA ERS statistics on fertilizer use (https://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx), covering the period of 1960-2018. The irrigation map used was reconstructed at a 1-km resolution based on the MODIS irrigated agriculture dataset for the United States (MIrAD-US) (Pervez and Brown, 2010).

### 4.2.3 Model Description

The agricultural module of the dynamic land ecosystem model (DLEM-Ag) is a highly integrated process-based agroecosystem model. The DLEM-Ag is capable of simulating the daily crop growth and exchanges of trace gases ($CO_2$, $CH_4$, and $N_2O$) between agroecosystems and the atmosphere; and quantifying fluxes and storage of carbon, water, and nitrogen within agroecosystem components as affected by multiple factors such as climate, atmospheric $CO_2$, nitrogen deposition, tropospheric ozone, land use and land cover change, and agriculture management practices (e.g., harvest, rotation, irrigation, and fertilizer use). This model has been extensively used to study crop production, SOC, exchanges of trace gases between agroecosystems and the atmosphere. The detailed structure and processes have been well documented in previous work (e.g., Tian et al., 2010; Ren et al., 2011; Ren et al., 2012; Ren et al., 2016; Zhang et al., 2018).

As we described in Huang et al. (2020), the implementation of tillage in DLEM-Ag focuses mainly on two processes directly affected by tillage: 1) the redistribution of surface residues with tillage practice and subsequent effects on soil water properties and water-related processes; 2) the increase in decomposition rates. The effect of tillage is implemented in combination with residue management, as these management practices are often interrelated (Strudley et al., 2008). Tillage incorporates surface residues into the soil, altering the coverage of residues on top of the soil. Crop residues left on soil surface intercept rainfall, facilitating water infiltration. Surface residues also serve as a barrier that lowers soil evaporation and reduces water losses to the atmosphere. Therefore, residues help maintain or improve soil moisture. Soil moisture affects primary production by regulating the amount of available water for plants, and *vice versa*. Soil moisture is also intimately associate with soil temperature.

### 4.2.4 Model Experiments Design

In this study, we designed three simulation experiments for assessing the magnitude and spatiotemporal patterns of crop yield and GHG emissions from 1980-2018, and for analyzing the difference caused by different tillage systems (Table 4.1). The model simulation began with an equilibrium run using 30-year (1980 -2009) mean climate datasets to develop the simulation baseline, in which the yearly variations of carbon, nitrogen, and water pools in each grid were less than 0.1 g C/m$^2$/yr, 0.1 mm $H_2O$/yr, and 0.1 g N m$^2$/yr, respectively. Before the transient run, the model was run for another 100 years for the spin-up to remove system fluctuations caused by the shift from equilibrium to transient mode, using climate data randomly selected from 1980-2008. The first simulation (S1) was designed to produce the near-real crop yield/GHG emissions and their changes in Kentucky, which was driven by historical varying tillage types and other input drivers (e.g., climate, $CO_2$, N deposition, fertilizer use, irrigation). In the second to third simulations (S2 – S3), we assumed that all the croplands were fixed under one tillage system since 1980 (Table 4.1).

Table 4.1 Experiments design in this study

| Experiments | Abbr | Drivers used | |
|---|---|---|---|
| | | Tillage | Others[a] |
| Historical varying tillage | S1 | 1980 - 2018 | Varying |
| Conventional tillage | S2 | 1980[b] | Varying |
| No tillage | S3 | 1980[c] | Varying |

Note: [a] Others include climate data (e.g., air temperature, precipitation, and radiation from 1980 to 2018), agricultural N fertilizer (i.e., N fertilizer from 1980 to 2018), and atmospheric conditions (i.e., $CO_2$ and N deposition from 1980 to 2018); [b] Tillage intensity across Kentucky for the entire period was consistent as conventional tillage (CT); [c] Tillage intensity across Kentucky for the entire period was consistent as no-tillage (NT).

## 4.3 Results and Discussion

### 4.3.1 Historical Climate Changes in Kentucky

The climate in Kentucky was generally becoming warmer and wetter from 1980 to 2018. Overall, the air temperature has been increasing at 0.02 °C/year ($R^2$=0.11, p=0.04) in Kentucky since 1980 (Figure 4.2c). The most rapid warming occurred in the west and east regions of Kentucky (Figure 4.2a). Similarly, the annual precipitation showed a significant increasing trend (7.05 mm/year, $R^2$=0.15, p=0.02) across the state (Figure 4.2d), with the most rapid wetting occurred in the north-central regions of Kentucky (Figure 4.2b). The decadal mean annual precipitation increased from about 1226 mm in the 1980s to 1448 mm in the 2010s. Most of the corn and soybean croplands located in western and central regions have experienced moderate warming and wetting climate compared to other areas. There were four relative droughts (large increase in temperature and decrease in precipitation) that occurred in 1987, 1999, 2005, and 2012 and six abnormally wet periods (large increase in precipitation and small change in temperature) in 1989, 1996, 2003, 2009, 2011, and 2018.

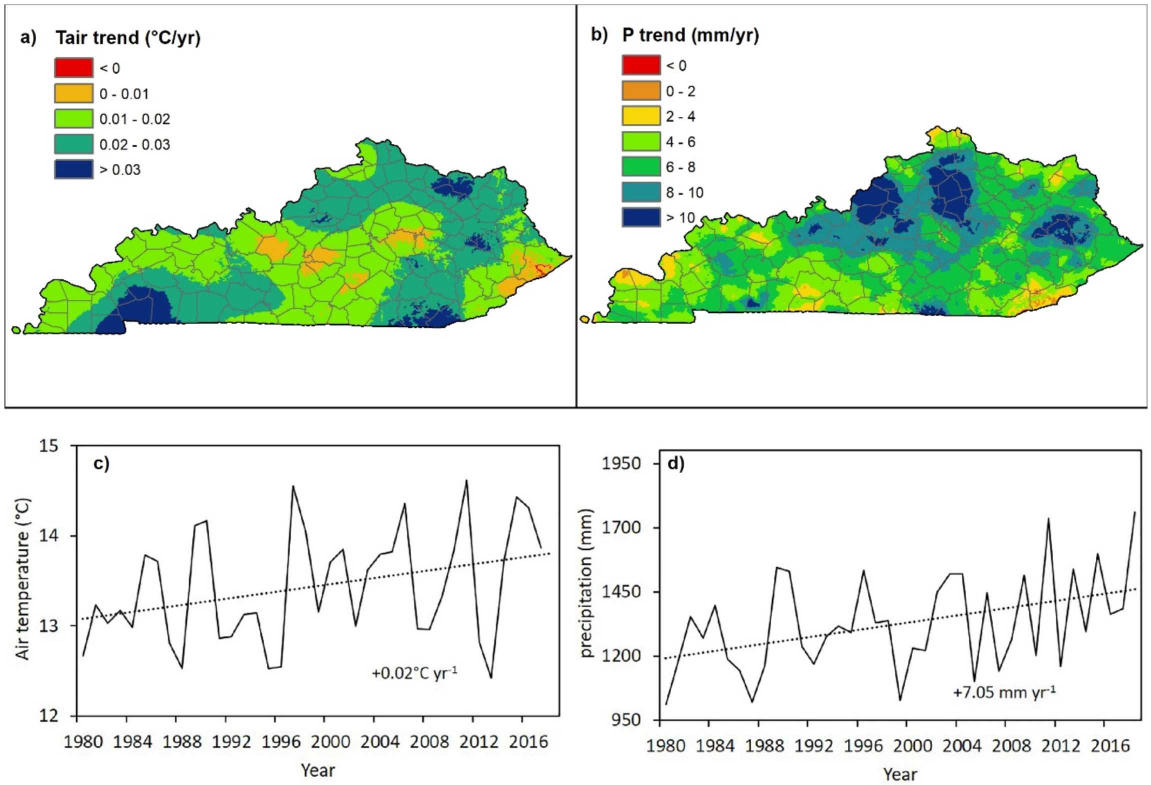


Figure 4.2 Spatial and temporal change in annual air temperature (a, c), and precipitation (b, d) (air temperature and precipitation trends are from 1980 to 2018)

### 4.3.2 Evaluation of DLEM-Ag Simulated Results

We first compared the model simulated crop yields against the reported USDA crop yields for corn and soybean at the state level from 1980 to 2018 (Figures 4.3 and 4.4a). The simulated results well captured the increasing trends in corn and soybean yields during the study period, with slightly lower rates than those from the USDA data (4.03 g C $m^{-2}$ $year^{-1}$ vs. 5.27 g C $m^{-2}$ $year^{-1}$ for corn, 1.17 g C $m^{-2}$ $year^{-1}$ vs. 1.65 g C $m^{-2}$ $year^{-1}$ for soybean). As presented in Figure 4.4a, the simulated yields by DLEM-Ag agreed well with the USDA crop yield ($R^2$=0.88). We also compared the simulated yields with the estimated yield from USDA inventory at the county level during 1980-2018. The county-level comparisons also showed high correlation coefficients ($R^2$=0.85). The calibration procedure used is responsible for this good agreement.

The simulated crop yields at the state level also well captured the temporal patterns in the survey data, with a correlation coefficient of 0.81 and 0.75 for corn and soybean, respectively (Figure 4.3). Although changes in production technology (improved hybrids and management practices) were mainly responsible for the upward trend in crop yields (Egli, 2008; Fischer et al., 2014), environmental factors, such as climate changes, were likely responsible for the annual variations (Hatfield et al., 2011). Some suggested that the increases in crop yield were also associated with changes in rainfall and temperature (Anderson et al., 2001; Lobell and Asner, 2003). The simulated results showed similar annual yield change pattern, but lower variance, compared to the survey data. The possible reason could be due to the uncertainties of climate data. In addition, the model simulation assumed that corn and soybean were planted in late-April in western Kentucky and early-May in central and eastern areas throughout the study period, which represented the optimal planting period of Kentucky (Lee et al., 2007). Compared to the large variation in planting dates in reality, the relatively stable and optimal planting dates could lead to less change in the simulated annual crop yields.

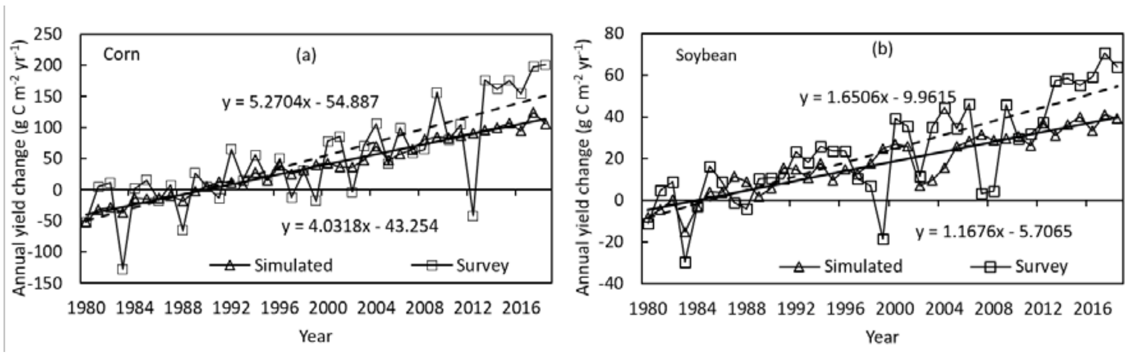


Figure 4.3 Changes in annual crop yield (relative to the average for 1980-1989) of Kentucky's corn (a) and soybean (b) estimated by DLEM-Ag model and USDA-NASS survey (the solid and dashed lines are linear trends for simulated and survey yield, respectively)

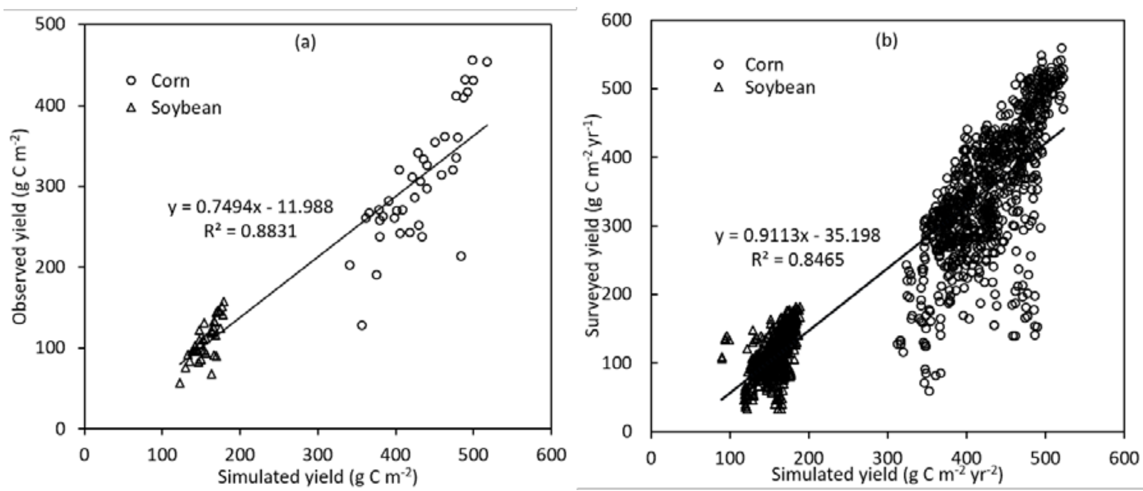


Figure 4.4 Comparison between the DLEM-Ag simulated crop yields and the USDA-NASS survey estimated crop yields for corn and soybean from 1980 to 2018 at (a) state- and (b) county- level. Counties are randomly selected

### 4.3.3 Tillage Effects on Crop Yield and GHG Emissions

In general, our simulations showed that adopting NT slightly increased corn yield (0.2%) and decreased soybean yield (-2.4%) on average (Figure 4.5a, b). These differences suggested that the yield differences between NT and CT were minimal in Kentucky, mostly due to the humid climate and medium- to well-drained farmland soils. In terms of tillage effects on GHG emission, our simulated results showed that $CO_2$ emissions under NT compared to CT were generally reduced by -1.6% for corn and by -4.53% for soybean (Figure 4.5c, d). Switching from CT to NT management decreased $N_2O$ emissions by an average of -10.49% for corn and by -19.64% for soybean (Figure 4.5e, f). The spatial patterns of tillage effects on GHG emissions were similar, with the relatively more substantial reduction in $CO_2$ and $N_2O$ emissions in the central and northern areas and relatively smaller reductions in the western regions.

The decrease in $CO_2$ emissions under the NT scenario compared to CT is consistent with previous findings (Abdalla et al., 2013; Lutz et al., 2019; Huang et al., 2020), as NT decreases the organic matter decomposition rates with less soil disturbance and lower soil temperature (Rastogi et al., 2002; Lu et al., 2016). However, the NT effects on reducing $CO_2$ emissions diminished with the duration of NT (Figure 4.6), suggesting that the soil and litter C stocks were increasing to enable rising $CO_2$ emissions under NT (Huang et al., 2018; Lutz et al., 2019), which gradually decrease the differences between tillage systems. Our results of reduced $N_2O$ emissions due to NT agrees with some previous studies (Omonode et al., 2011; Yoo et al., 2016; Plaza-Bonilla et al., 2018) but contradicts several literature studies (Huang et al., 2018; Mei et al., 2018; Lutz et al., 2019). The reduction in $N_2O$ emissions may be due to the well-aerated farmland soils (Rochette 2008). The sequential nitrification and denitrification that are responsible for $N_2O$ emissions were in the optimal soil temperature and moisture conditions under NT (Doran 1980; Williams et al., 1992). Higher levels of inorganic N could lead to higher $N_2O$ emissions. However, N mineralization rates are often lower under NT than under CT due to the leaving of crop residues on the soil surface (Rice et al., 1986; Franzluebbers et al., 1995; Dick et al., 2008).

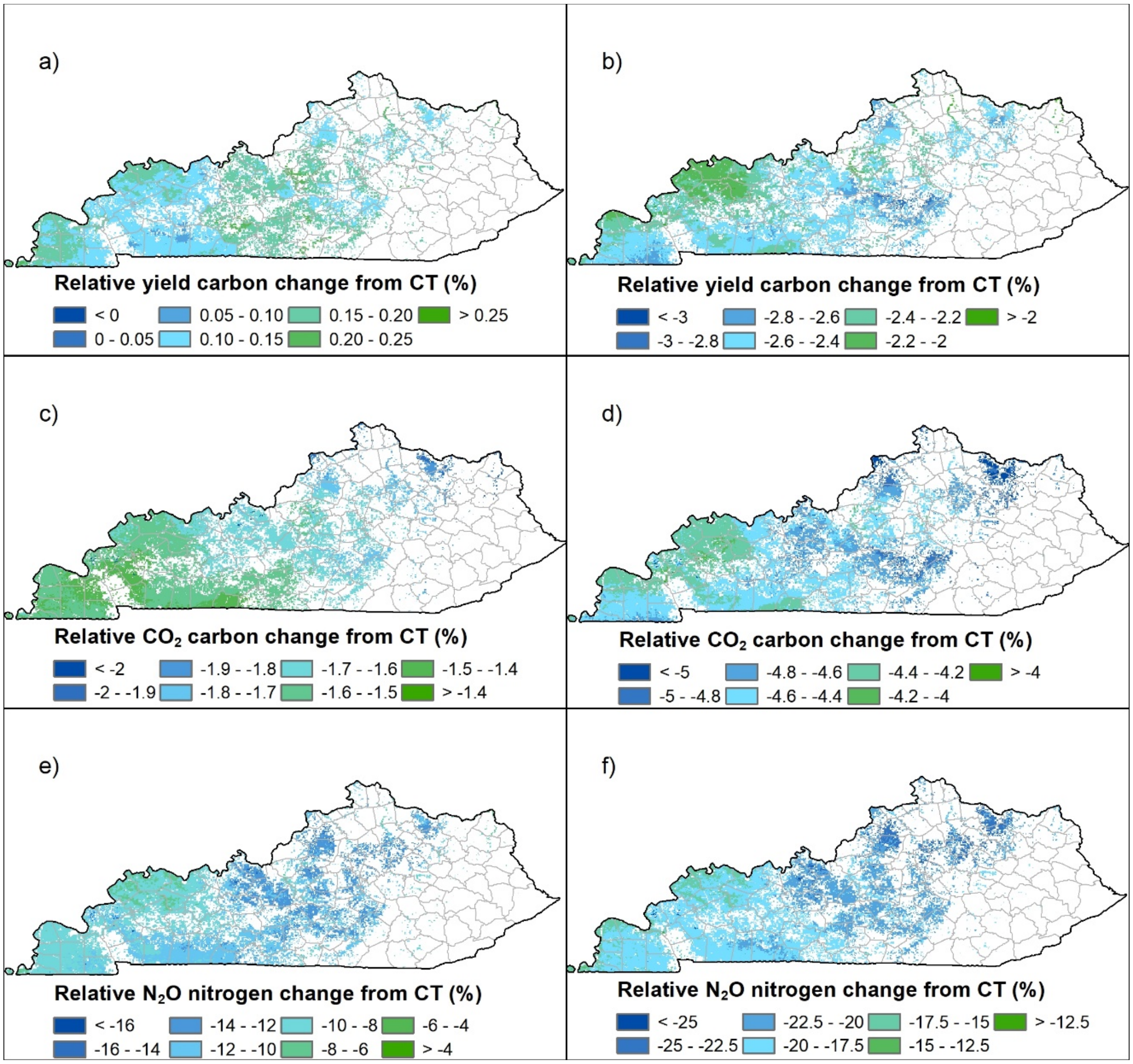


Figure 4.5 Relative changes in yield-C (a, b; carbon content of grain yield), soil $CO_2$-C (c, d; carbon content of $CO_2$) and $N_2O$-N (e, f; nitrogen content of $N_2O$) emissions for NT vs. CT comparisons for corn (left panel) and soybean (right panel)

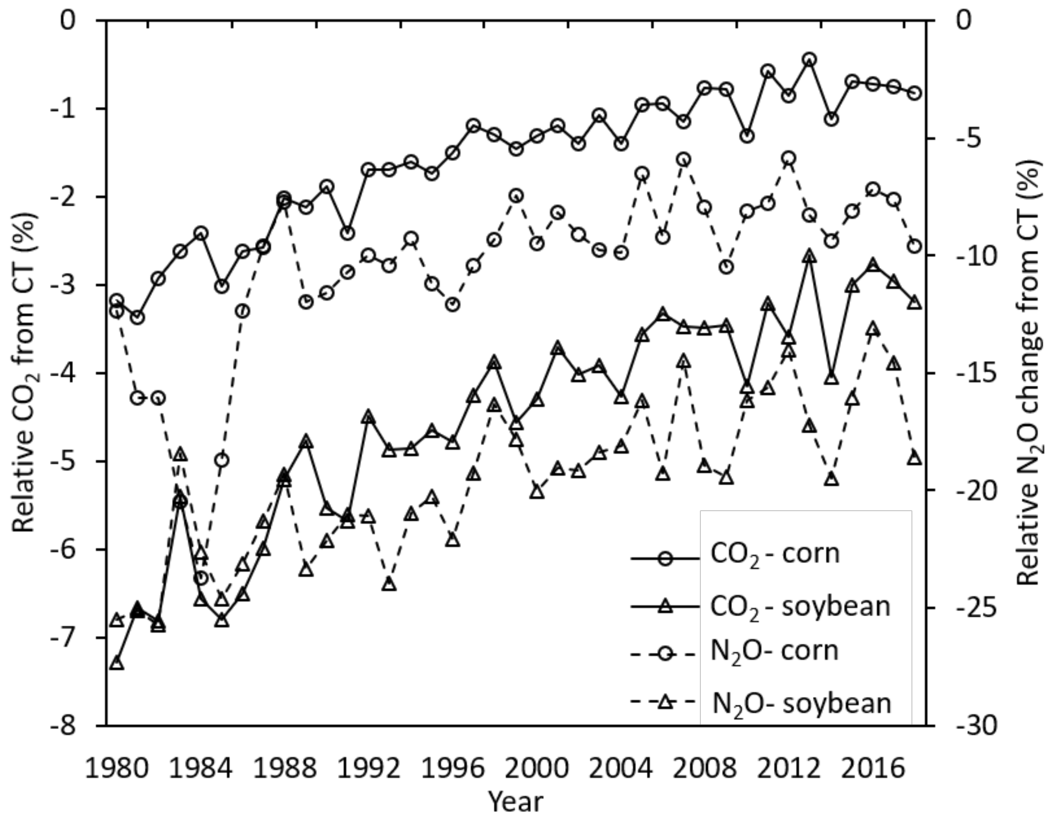


Figure 4.6 Relative changes in $CO_2$ and $N_2O$ for NT vs. CT comparisons for corn and soybean during 1980-2018

### 4.3.4 Climate and Tillage Effects on GHG Emissions

We evaluated tillage effects on GHG emissions as affected by spatial climate characteristics (i.e., annual precipitation and temperature) across Kentucky for corn and soybean croplands. Generally, annual precipitation did not significantly influence the tillage effects on either $CO_2$ and $N_2O$ emissions in Kentucky corn and soybean croplands (Figure 4.7a, c, e, g). In comparison, the differences in GHG emissions between NT and CT tended to decrease with increasing annual temperature in both cropping systems (Figure 4.7b, d, f, h). Such correlations were more pronounced for changes in $CO_2$ emissions than those in $N_2O$ emissions. The results suggest that spatial air temperature pattern, but not precipitation, is a dominant factor affecting the spatial heterogeneity in NT effects on GHG emissions in Kentucky. No-tillage can lead to more reduction in GHG emissions in cooler (i.e., northern Kentucky) than warmer (western and southern Kentucky) regions. Our results agreed with a recent study in northern Kentucky that NT effects on SOC were more controlled by temperature than precipitation (Huang et al., 2020), as soil GHG emissions and SOC are highly correlated.

### 4.3.5 Soil Texture and Tillage Effects on GHG Emissions

Our simulation results showed that the relative $N_2O$ changes for NT vs. CT comparisons significantly increased with the increasing clay content in soils (Figure 4.8g, j), but tended to decrease with rising sand and silt contents (Figure 4.8h-i, k-l). In comparison, the differences in $CO_2$ emissions between NT and CT management were not significantly affected by soil texture (Figure 4.8a-f). Clay content in the soil is strongly correlated with SOC (Meersmans et al., 2012). By binding organic matter, clay particles help form and stabilize soil aggregates, imposing a physical barrier between decomposer and organic substrates (Dominy et al., 2002). Compared to $CO_2$, the production of $N_2O$ in the soil is more sensitive to soil clay content.

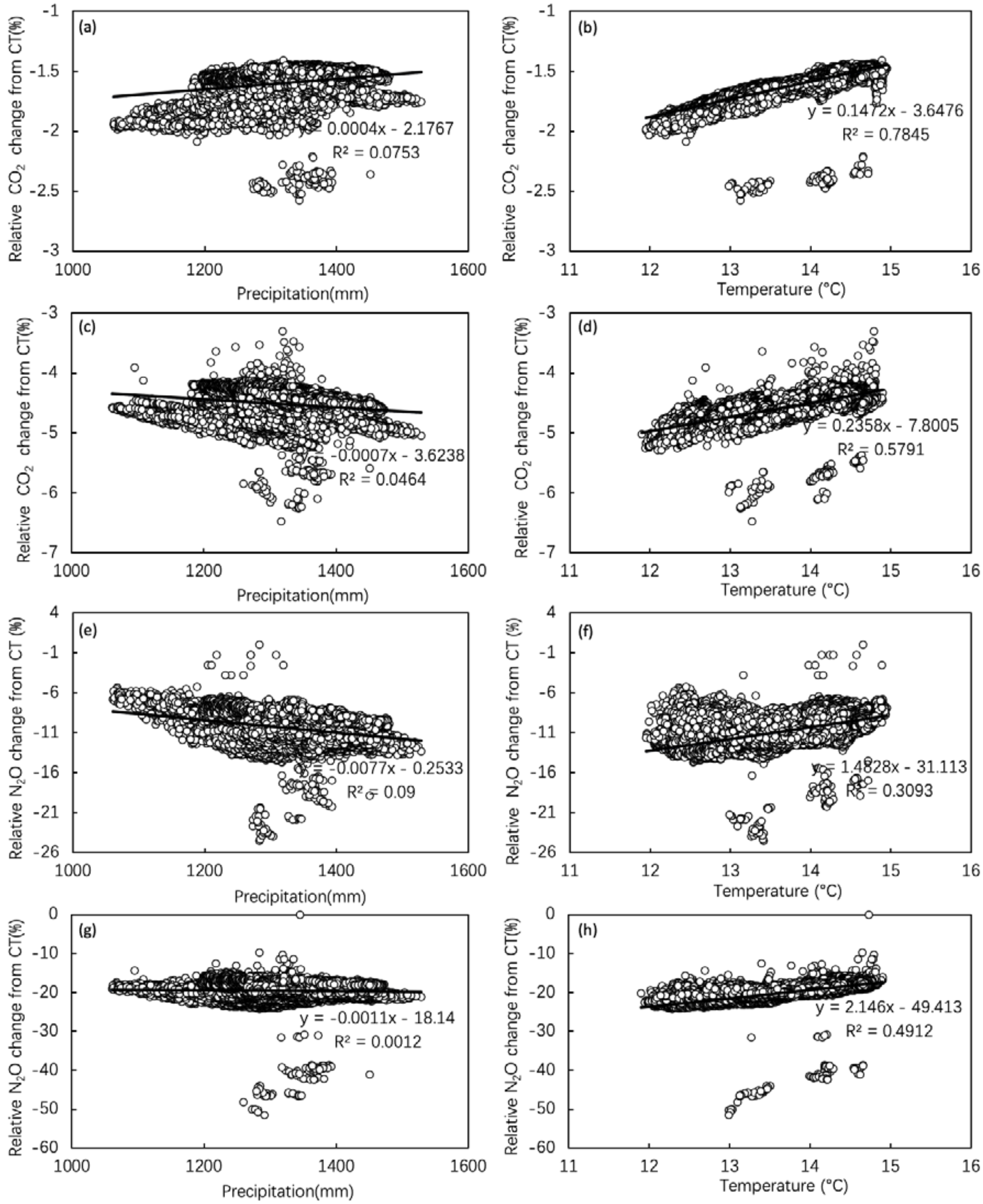


Figure 4.7 Linear regression of relative $CO_2$ emission (a-d) and $N_2O$ emission (e-h) changes for NT vs. CT comparisons on annual precipitation (left panel) and temperature (right panel). (a, b, e, f) are for corn and (c, d, g, h) are for soybean

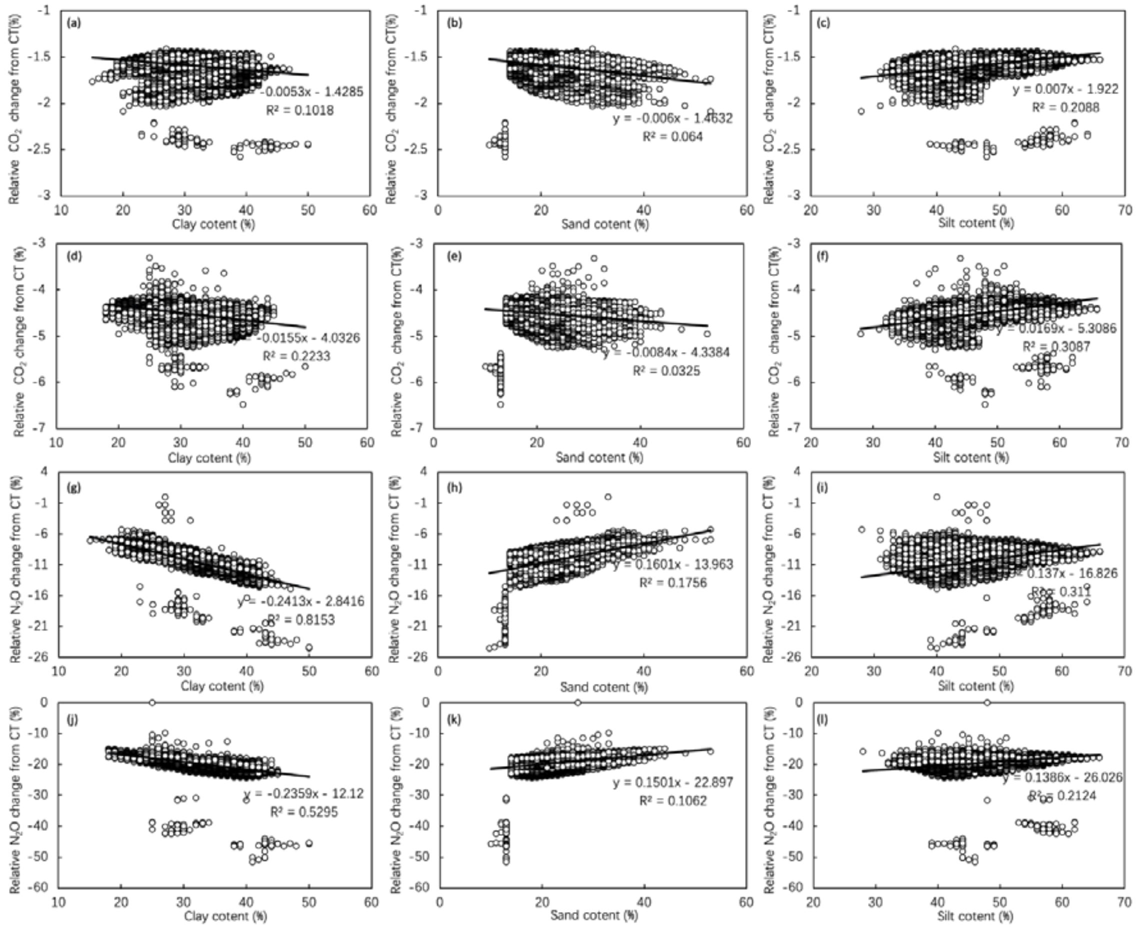

Figure 4.8 Linear regression of relative $CO_2$ emission (a-f) and $N_2O$ emission (g-l) changes for NT vs. CT comparisons on clay content (left panel), sand content (middle panel), and silt content (right panel). (a, b, c, g, h, i) are for corn and (d, e, f, j, k, l) are for soybean

## 4.4 Conclusions

This study provided the first attempt to quantify the effects of NT on crop yield and soil GHG emissions at the regional scale by using an agroecosystem modeling approach. Overall, the model showed reasonable performance in Kentucky. By conducting the sensitivity simulation, we found that NT had no significant effect on corn and soybean yield but decreased soil $CO_2$ emissions (-1.6% for corn and -4.53% for soybean) and $N_2O$ emissions (-10.49% for corn and -19.64% soybean) compared to CT in Kentucky.

Temperature and soil clay content are the two important factors that govern the effectiveness of NT in reducing soil GHG emissions. The increasing temperature would lead to less NT benefit in decreasing soil GHG emissions. There is a tendency that NT reduces soil $N_2O$ emissions more in high clay content soils in Kentucky. Finally, our work gives some evidence that NT is a useful option in climate change adaptation and mitigation in Kentucky.

## CHAPTER 5. SIMULATING THE EFFECTS OF TILLAGE ON CROP WATER PRODUCTIVITY IN CORN AND SOYBEAN SYSTEMS ACROSS THE OHIO RIVER BASIN

### 5.1 Abstract

Improvement in management practices is imperative for building agroecosystems' resilience and climate change adaptation and for ensuring food security and agriculture sustainability in the long term. The past half-century has witnessed a revolution in tillage methods (i.e., conservation tillage) to combat soil erosion and degradation in the United States. However, its effects on crop productivity and water use patterns remain controversial from local trials, with more uncertainty from the macro-scale perspective. Here, we used a process-based agroecosystem model in combination with spatial-explicit gridded data to quantify the long-term effects of conservation tillage (e.g., no-tillage (NT) and reduced tillage (RT)) on crop water productivity (CWP = crop production / evapotranspiration) of corn and soybean in the Ohio River Basin during 1979-2018. We found an average of 4.76% and 5.87% CWP increase for corn and soybean, respectively, if all the fields employed NT treatment. When compared to the conventional tillage scenario, NT and RT both would enhance CWP, primarily due to reduced evapotranspiration under NT and RT scenarios. Simulation results showed that although NT and RT reduced surface runoff, they could potentially increase subsurface drainage and nutrient leaching from corn and soybean farmland. Our study demonstrates that along with conservation tillage, water and nutrient management should be further considered to enhance soil water retention and nutrient use in the study region. Our findings also provide insight into optimizing management practices for other areas where conservation tillage is widely applied.

## 5.2 Introduction

Water deficits and surpluses are the primary perils to agriculture worldwide, especially in rain-fed regions (Shekhar and Shapiro, 2019). As water competition between cities and agriculture increases and climate change exacerbates water stresses (Brauman et al., 2013; Drum et al., 2017), there has been an ongoing search for strategies to maximize crop yield and biomass for each drop of water in agriculture. The key to achieving this goal is to enhance crop water productivity (CWP), which is defined as the ratio of crop carbon gain (e.g., gross primary production) to water consumption (e.g., evapotranspiration). Crop water productivity couples the carbon and water cycles in the agroecosystem (van Halsema and Vincent, 2012) and is a vital indicator in the evaluation of agricultural performance.

Conservation tillage is a promising practice that helps conserve soil moisture and reduce soil erosion, thus helping crops under water stress (Busari et al., 2015; Holland 2004; Phillips et al., 1980). Conservation tillage is any tillage system with a seedbed preparation technique in which at least 30% of the soil surface is covered by crop residue (Lal et al., 2017), including no-tillage (NT), reduced tillage (RT), mulch tillage, and ridge tillage. Compared to conventional tillage (CT), conservation tillage can lead to less soil disturbance and more surface residues. The advantage of conservation tillage in improving CWP has been widely reported across different agroecosystems (Cantero-Martínez et al. 2007; Jabro et al., 2014; Li et al., 2018; Su et al., 2007; Tang et al., 2015). However, some studies argue that conservation tillage led to negligibly different (Irmak et al., 2019) or lower (Guan et al., 2015; Liu et al., 2013) CWP than CT. Tillage effects on CWP may vary in several ways, depending on tillage type and duration, as well as its interaction with a different climate, soil, management, and cropping systems (Strudley et al., 2008). Although it is essential to assess the suitability and effectiveness of conservation tillage prior to its implementation at local scales, there is an urgent need to evaluate its performance at the macroscale, considering its widespread use in the past several decades.

Field observations, including eddy covariance methods (Chi et al., 2016; Lampurlanés et al., 2016; Schwartz et al., 2019; Tang et al., 2015), have been used to estimate CWP at specific sites accurately. However, new efforts are needed to determine the spatial heterogeneity of CWP. Besides, previous CWP studies tended to focus more on inefficiency under arid/semi-arid conditions and neglect evaluation of how well conservation tillage affects crop water use in humid conditions. Additionally, remote sensing products (e.g., MODIS GPP and ET) are often used to quantify large-scale CWP (Ai et al., 2020; Lu and Zhuang; 2010). However, these products cannot provide crop-specific CWP because they do not consider crop spatial distribution. Previously, regional and global CWP simulations generally ignored the key anthropic factor of tillage, in part because of the under-representation of tillage in global ecosystem models (Lutz et al., 2019). It is critical to integrate crop type and tillage distribution into ecosystem models to simulate the spatiotemporal CWP of different crops accurately.

The Ohio River Basin (ORB) is in the Eastern Corn Belt of the U.S., and almost 98% of its cropland is cultivated with corn and soybean according to the 2018 National Cropland Data Layer. The ORB is one of the earliest regions in the world to implement conservation tillage (e.g., NT and RT). Specifically, the adoption of conservation tillage systems has been continually increasing during the past several decades. As of 2018, more than 60% of corn and almost 80% of soybean in the ORB were under different forms of conservation tillage (CTIC, 2018). The widespread alterations in tillage systems in the

ORB justifies the need for scientific investigation of the impacts of these practices on water resources, including evapotranspiration, productivity, and CWP. However, knowledge gaps still exist regarding the long-term and spatial-explicit effects of different tillage practices on CWP in the corn and soybean cropping systems of the ORB region. Here we used a process-based model (DLEM-Ag) to quantify the magnitude and spatial-temporal pattern of CWP in the ORB croplands during 1979-2018. Specific objectives were to: 1) investigate the magnitude and long-term trend in CWP; 2) quantitatively examine the changes in CWP as affected by different tillage systems in the ORB corn and soybean cropping systems.

## 5.3 Materials and Methods

### 5.3.1 Description of the Study Area

The Ohio River Basin (ORB) covers 421,966 km$^2$ within 11 states. The Ohio River starts at the confluence of the Allegheny and the Monongahela in Pittsburgh, Pennsylvania, and ends in Cairo, Illinois, where it flows into the Mississippi River. The humid continental climate is prevalent in the upper half of the basin, and a humid subtropical climate is dominant in the lower half of the basin. The whole ORB receives a high amount of rainfall, with an average annual rainfall of 1250 mm. About half of the land cover in this basin is forested land, primarily of deciduous trees. Cultivated cropland (~ 30%) is dominant on the northern side of the Ohio River and the western part of the basin. Corn and soybean are the major crops grown here (Santhi et al., 2014).

### 5.3.2 Input Driving Data

#### 5.3.2.1 Climate, $CO_2$, and Nitrogen Deposition Data

The daily climate data used to drive the model were derived from the gridMET dataset at a resolution of 4 km × 4 km covering the United States from 1979-2018 (Abatzoglou 2013), including maximum, minimum, and average temperature; precipitation; shortwave radiation; wind; and relative humidity. The historical $CO_2$ concentration dataset was retrieved from NOAA/GML (www.esrl.noaa.gov/gmd/ccgg/trends/). Gridded N deposition maps were adapted from the North American Climate Integration and Diagnostics – Nitrogen Deposition Version 1 (NACID-NDEP1) dataset (Hember 2018).

#### 5.3.2.2 Crop Rotation Map and Crop Phenology Data

The crop rotations used in the ORB were created by using the USDA-NASS Cropland Data Layer (CDL) datasets. Following a similar approach by Panagopoulos et al. (2015) and Srinivasan et al. (2010), we overlaid multi-years of CDL information to produce crop rotation maps. The 2018 CDL data showed that approximately 98% of the ORB cropland was cultivated with corn and soybean. Based on a three-year rotation pattern in the ORB from 2015-2017, we derived eight cropland rotation types involving corn and soybean: 1) corn/soybean, 2) corn/soybean/soybean, 3) corn/corn/soybean, 4) soybean/corn, 5) soybean/corn/corn, 6) soybean/soybean/corn, 7) continuous corn, and 8) continuous soybean. These eight rotation types constitute approximately 90% of all the three-year rotations that involve corn or soybean in the ORB. Thus, we assumed that minor rotations at each 30-m pixel, such as corn/soybean/wheat or corn/corn/wheat, were eliminated and replaced with one of the eight rotations within the nearest pixel. The percentages of corn and soybean at each 4-km pixel were then calculated based on the 30-m layers.

The phenology of corn and soybean were derived by using the crop phenology dataset from Yang et al. (2020) in combination with the CDL datasets. Information from the phenology dataset includes the planting and harvesting date of crops. Specifically, we: 1) calculated the percentage of corn and soybean, respectively, at each 500-m pixel to match the phenology dataset; 2) overlaid the central coordinate of each 4-km pixel on the 500-m phenology map to assign the index of the 500-m pixel to the nearest 4-km pixel; 3) searched within 10 km around the central coordinate on the 4-km map to find the pixels with more than 55% of corn or soybean; 4) assigned the planting/harvesting date of corn and soybean at the nearest pixel to the central coordinated of the 500-m pixel. For unassigned pixels, we replaced the value with the most adjacent pixels. Overall, the planting dates in the ORB were from 97-177 (day of the year) for corn and soybean. The harvesting dates were from 289-330 and 277-290 for corn and soybean, respectively.

#### 5.3.2.3 Tillage and Other Agricultural Management Practices

We obtained county-level ORB tillage information from the Conservation Technology Information Center's (CTIC; https://www.ctic.org/) National Crop Residue Management Survey (CRM). The tabular data provides the acreages and percentages of five tillage types adopted in all crops, including corn and soybean. For simplification, we grouped the five major tillage types into three categories, i.e., no-tillage (NT), reduced tillage (RT, including ridge tillage, mulch tillage, and reduced tillage), and conventional tillage (CT). We used county acreages in combination with the CDL-derived cropland layer to estimate the spatial distribution of conventional and conservation tillage percentages for corn and soybean, assuming each pixel within a county has the same percentages of tillage-specific area. We reconstructed annual tillage maps from 1989-2011 based on the CRM dataset and made the assumptions that the tillage maps of other years are similar to the nearest year. Moreover, we also generated three ideal tillage maps with all the corn/soybean under a unique tillage regime (be it NT, RT, or CT) for sensitivity analysis.

Crop specific N fertilizer use data were derived from the USDA ERS statistics on fertilizer use (https://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx), covering 1960-2018. An irrigation map was reconstructed at a 4-km resolution based on the MODIS irrigated agriculture dataset (2012) for the United States (MIrAD-US, Pervez and Brown, 2010).

### 5.3.3 Model Description

#### 5.3.3.1 The DLEM-Ag

The agricultural module of the dynamic land ecosystem model (DLEM-Ag) is a highly integrated process-based agroecosystem model. The DLEM-Ag is capable of simulating the daily crop growth and exchanges of trace gases ($CO_2$, $CH_4$, and $N_2O$) between agroecosystems and the atmosphere; and quantifying fluxes and storage of carbon, water, and nitrogen within agroecosystem components as affected by multiple factors such as climate, atmospheric $CO_2$, nitrogen deposition, tropospheric ozone, land use and land cover change, and agriculture management practices (e.g., harvest, rotation, irrigation, and fertilizer use). This model has been extensively used to study crop production, SOC, exchanges of trace gases between agroecosystems and the atmosphere. The detailed structure and processes have been well documented in previous work (e.g., Tian et al., 2010; Ren et al., 2011; Ren et al., 2012; Ren et al., 2016; Zhang et al., 2018).

#### 5.3.3.2 Model Representation of Tillage Impacts

As described in Huang et al. (2020), the implementation of tillage in DLEM-Ag mainly focuses on two processes that are directly affected by tillage: 1) the redistribution of surface residues with tillage practice and subsequent effects on soil water properties and water-related processes; 2) the increase in decomposition rates. The effect of tillage is implemented in combination with residue management, as these management practices are often interrelated (Strudley et al., 2008). Tillage incorporates surface residues into the soil, altering the coverage of residues on top of the soil. Crop residues left on soil surface intercept rainfall, facilitating water infiltration. Surface residues also serve as a barrier that lowers soil evaporation and reduces water losses to the atmosphere. Therefore, residues help maintain or improve soil moisture. Soil moisture affects primary production by regulating the amount of available water for plants, and *vice versa*. Soil moisture is also intimately associate with soil temperature.

### 5.3.4 Model Evaluation

In the previous studies, the DLEM-Ag model has been extensively calibrated and validated against both site-level and regional-scale data. More details can be found in published studies (Ren et al., 2011, 2012, 2015; Tian et al., 2010; Zhang et al., 2018). Given that we used different model driving force from previous regional studies and we mainly focus on corn and soybean systems, we specifically calibrated and validated the simulated crop GPP and ET against published results from cropland sites in the AmeriFlux Network in and close to the ORB region. Generally, the model simulated GPP and ET showed good agreement with the measurements at flux towers (Figure 5.1a, b).

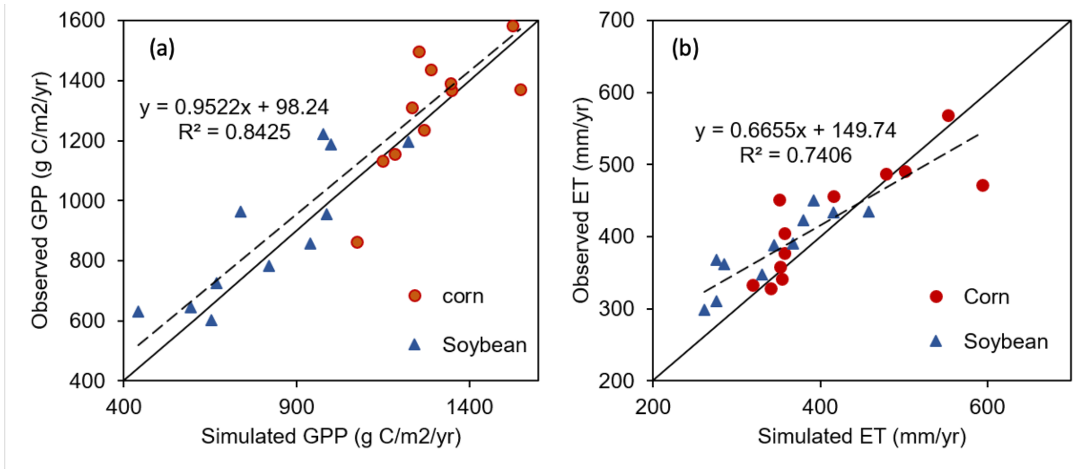


Figure 5.1 Comparison of the model estimated and observed gross primary productivity (GPP; a) and evapotranspiration (ET; b) for corn and soybean in and near the ORB region (dashed line is the regression of observed data and modeled results. The solid line is the 1:1 line)

### 5.3.5 Model Experiments Design

In this study, we designed four simulation experiments for assessing the magnitude and spatiotemporal patterns of corn and soybean CWP during 1979-2018, and for analyzing the difference caused by different tillage systems (Table 5.1). The model simulation began with an equilibrium run using 30-years (1979 -2008) mean climate datasets to develop the simulation baseline, in which the year-to-year variations of carbon, nitrogen, and water pools in each grid were less than 0.1 g C/m$^2$/yr, 0.1 mm $H_2O$/yr, and 0.1 g N m$^2$/yr, respectively. Before the transient run, the model was run for another 100 years for the spin-up to remove system fluctuations caused by the shift from equilibrium to transient mode, using climate data randomly selected from 1979-2008. The first simulation (S1) was designed to produce CWP that close to reality and its changes in the ORB, which was driven by historically varying tillage types and other input drivers (e.g., climate, $CO_2$, N deposition, fertilizer use, irrigation, and crop rotation). In the second to fourth simulations

(S2 – S4), we assumed that all the croplands were fixed under one tillage system since 1979. A comparison of four experiments provides the potential CWP change of adopting conservation tillage in the ORB corn and soybean systems.

Table 5.1 Experiments design in this study

| | | Drivers used | |
|---|---|---|---|
| Experiments | Abbr | Tillage | Others[a] |
| Historical varying tillage | S1 | 1979 - 2018 | Varying |
| Conventional tillage | S2 | 1979[b] | Varying |
| Reduced tillage | S3 | 1979[c] | Varying |
| No-tillage | S4 | 1979[d] | Varying |

Note: [a] Others include climate data (e.g., air temperature, precipitation, and radiation from 1979 to 2018), agricultural N fertilizer (i.e., N fertilizer from 1979 to 2018), and atmospheric conditions (i.e., $CO_2$ and N deposition from 1979 to 2018); [b] Tillage intensity across the ORB for the entire period was consistent as conventional tillage (CT); [c] Tillage intensity across the ORB for the entire period was consistent as reduced tillage (RT); [d] Tillage intensity across the ORB for the entire period was consistent as no-tillage (NT).

## 5.4 Results

### 5.4.1 Historical Changes in Air Temperature and Precipitation in the ORB

The ORB has experienced substantial changes and variability in climate (i.e., temperature and precipitation) during 1979-2018. A warming trend dominated the entire study area, with the most rapid warming occurring in the periphery of the ORB region, including western Kentucky, southern and eastern Indiana, and western Ohio (Figure 5.2a). Over the entire ORB region, air temperature has been increasing at 0.02 °C/year ($R^2 = 0.16$, $p < 0.05$) since 1979 (Figure 5.2b). In comparison, a wetting trend dominated the entire ORB with the most rapid precipitation increase occurring in the center of the ORB, along both sides of the middle Ohio River, especially in southeastern Indiana and northern/eastern Kentucky (Figure 5.2c). The average precipitation increased at 3.9 mm/year ($R^2$ = 0.10, $p$ < 0.05) since 1979 (Figure 5.2d). There were two severe droughts (large increase in temperature and decrease in precipitation) that occurred in 1987 and 2012 and two abnormally wet periods (large increases in precipitation with small changes in temperature) in 1996 and 2018.

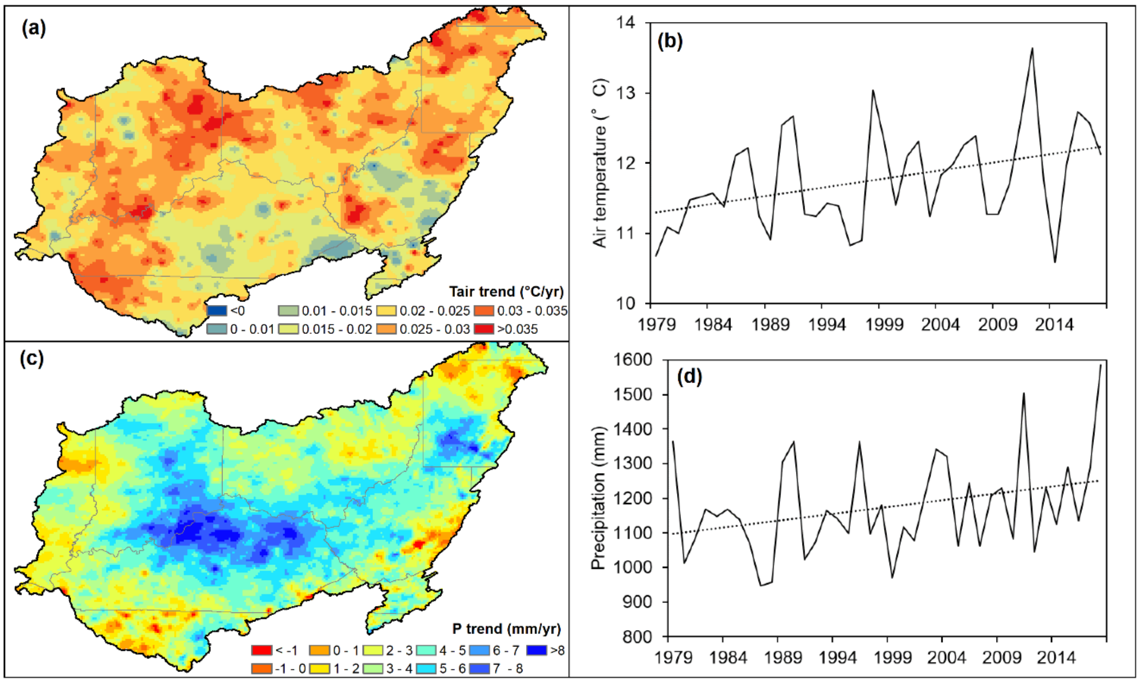


Figure 5.2 Spatial and temporal change of annual (a, b) air temperature, (c, d) precipitation (air temperature and precipitation trends are from 1979 to 2018)

### 5.4.2 Tillage Effects on GPP and ET over the ORB Region

In the ORB region, the mean annual GPP was 1091± 144g C/m$^2$/yr and 747 ± 199 g C/m$^2$/yr for corn and soybean, respectively (Figure 5.3a, b). The spatial distribution patterns of GPP for corn and soybean are similar to each other, with higher GPP in the northwest of the ORB region, where the ORB's primary croplands are located. Compared to the baseline simulation (S1), tillage scenario experiments (S2, S3, and S4) showed that the effect of tillage on GPP was negligible for both crops (Figure 5.3 c-h). Nevertheless, NT and RT tended to have a slight positive effect on GPP relative to CT.

The spatial distribution patterns of annual ET showed an increasing trend from the northeast toward the southwest for both crops in the ORB region (Figure 5.4a, b), with the average annual ET of 601 ± 37 mm/yr for corn and 508 ± 33 mm/yr for soybean. The sensitivity scenario experiments showed that CT increased corn ET by 2.8 ± 1.2% and soybean ET by 7.2 ± 2.4% (Figure 5.4c, d), while NT decreased corn ET by 3.9 ± 1.7% and soybean ET by 5.3 ± 3.1% (Figure 5.4g, h), compared to the baseline experiment (S1). The effect of RT on ET relative to S1 was somewhat neutral (-0.6 ± 1.4% and 1.1 ± 2.2% for corn and soybean, respectively, Figure 5.4e, f).

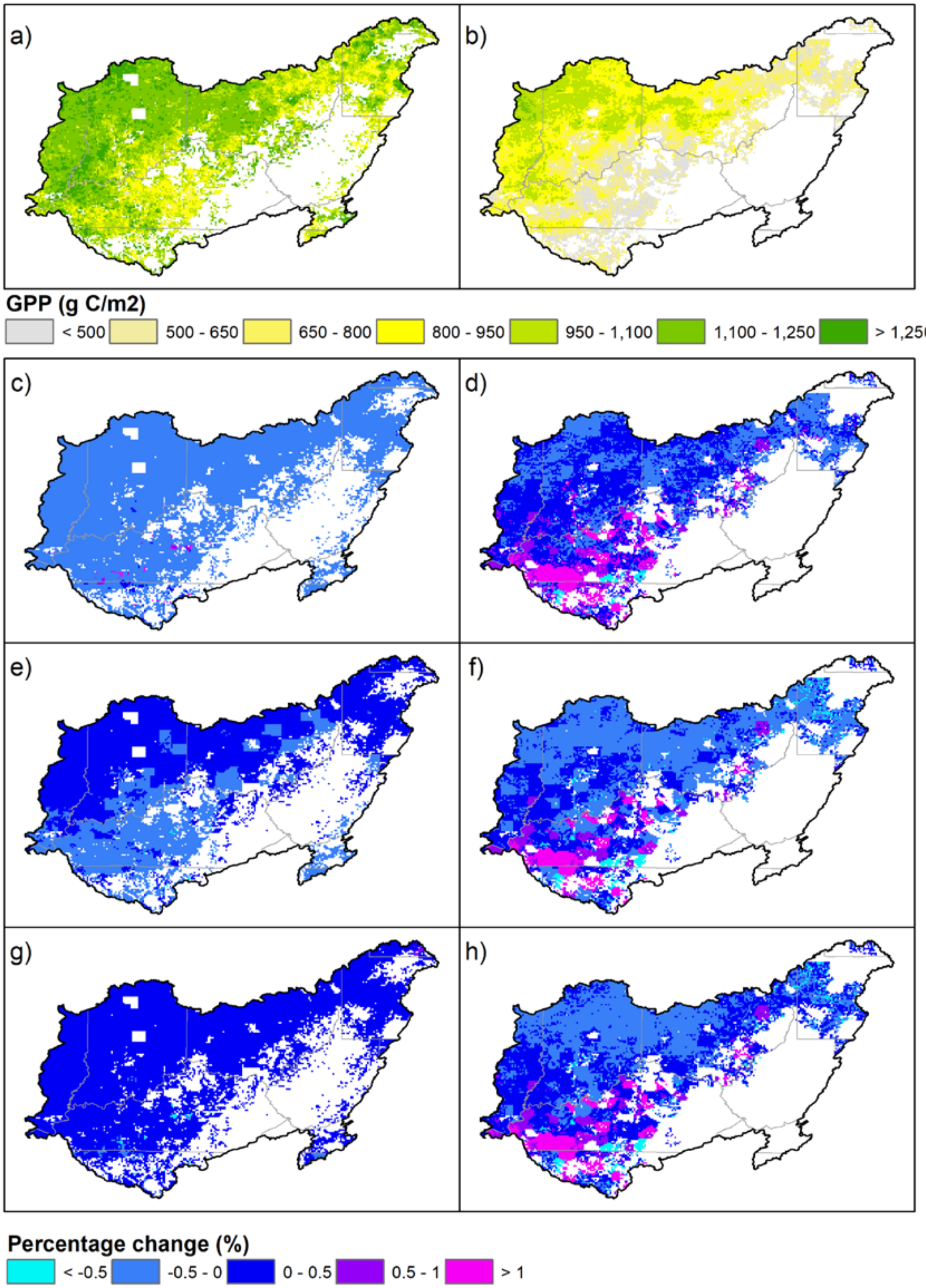


Figure 5.3 Spatial distribution of the mean annual (1979 - 2018) gross primary productivity (GPP) in the ORB region (a, b), and the percentage change from the baseline GPP owing to CT (c, d), RT (e, h), and NT (g, h) (Left panel is for corn and right panel is for soybean)

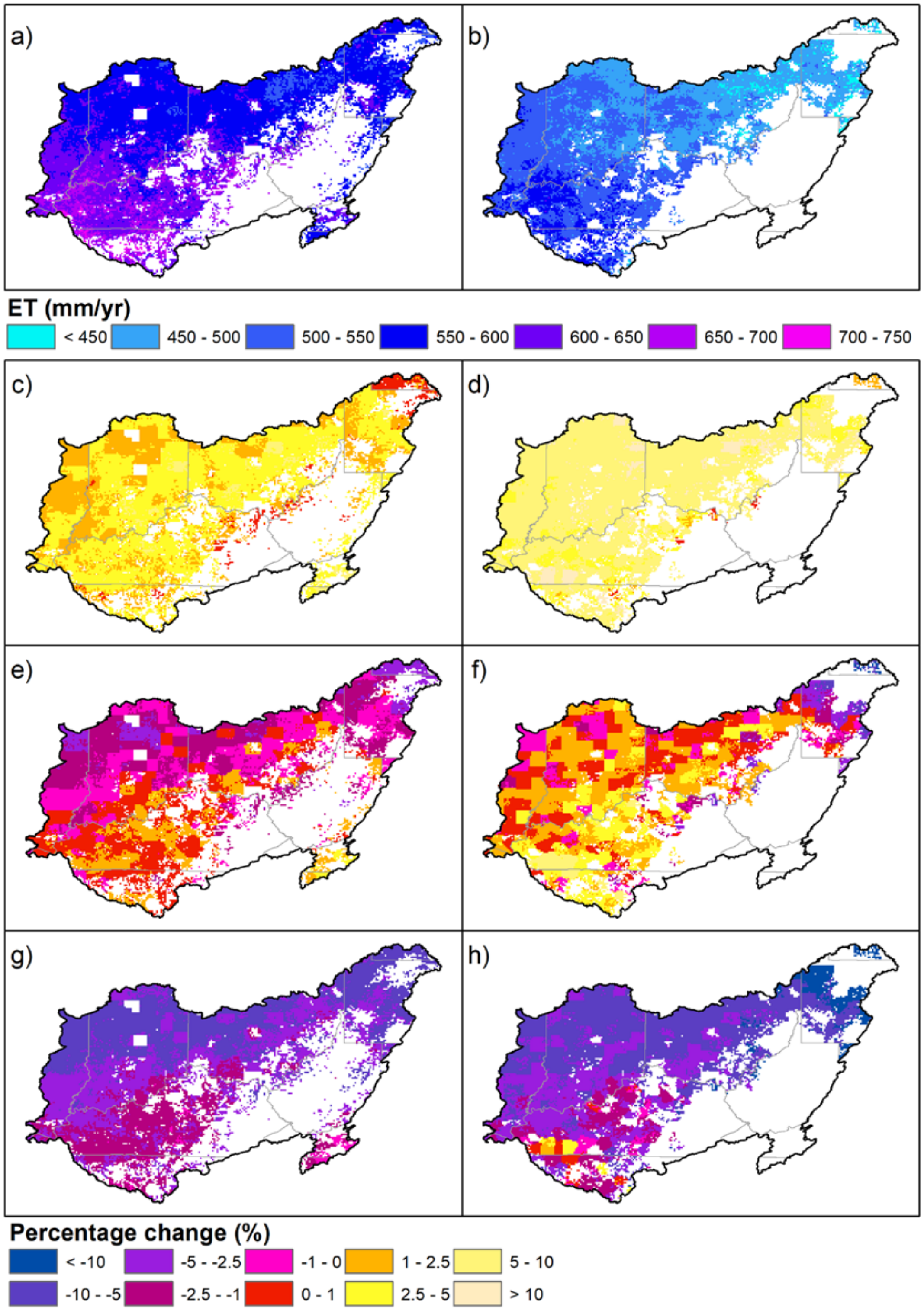


Figure 5.4 Spatial distribution of the mean annual (1979 - 2018) evapotranspiration (ET) in the ORB region (a, b), and the percentage change from the baseline ET owing to CT (c, d), RT (e, h), and NT (g, h) (Left panel is for corn and right panel is for soybean)

### 5.4.3 Tillage Effects on CWP over the ORB Region

The baseline simulation (S1) showed that the mean annual CWP was 1.79 ± 0.24 kg C/$m^3$ and 1.44 ± 0.37 kg C/$m^3$ for corn and soybean, respectively, across the ORB region during 1979 - 2018 (Figure 5.5a, b). The spatial patterns for the annual CWP were similar between corn and soybean. Higher CWP areas occurred in the northwest ORB and decreased southeastward. The sensitivity simulations (S2, S3, and S4) revealed that the tillage-induced CWP change varied among different tillage scenarios. Compared to the baseline experiment (S1), CT decreased the mean annual CWP by 2.95 ± 1.09% for corn and 6.41 ± 1.77% for soybean (Figure 5.5c, d), while NT increased CWP by 4.76 ± 2.28% and 5.87 ± 3.24% for corn and soybean, respectively (Figure 5.5g, h). However, the impact of RT on CWP was relatively neutral (0.50 ± 1.42% and -1.01 ± 1.92% for corn and soybean, respectively, Figure 5.5e, f).

The baseline temporal dynamics of the annual CWP showed that there was a significant increasing trend for soybean (0.006 kg C/$m^3$/yr, $p < 0.05$, Figure 5.6b) but not for corn (0.003 kg C/m3/yr, $p = 0.12$, Figure 5.6a). Generally, throughout the simulation period, the NT scenario resulted in the highest annual CWP for both crops in the ORB region (1.88 ± 0.12 kg C/$m^3$ and 1.50 ± 0.17 kg C/$m^3$ for corn and soybean, respectively). In comparison, the CT experiment led to the lowest annual CWP (1.74 ± 0.17 kg C/$m^3$ and 1.32 ± 0.21 kg C/$m^3$ for corn and soybean, respectively, Figure 5.6a, b), despite the variations in the annual CWP. No significant difference in the annual CWP was observed between the RT and the baseline experiments.

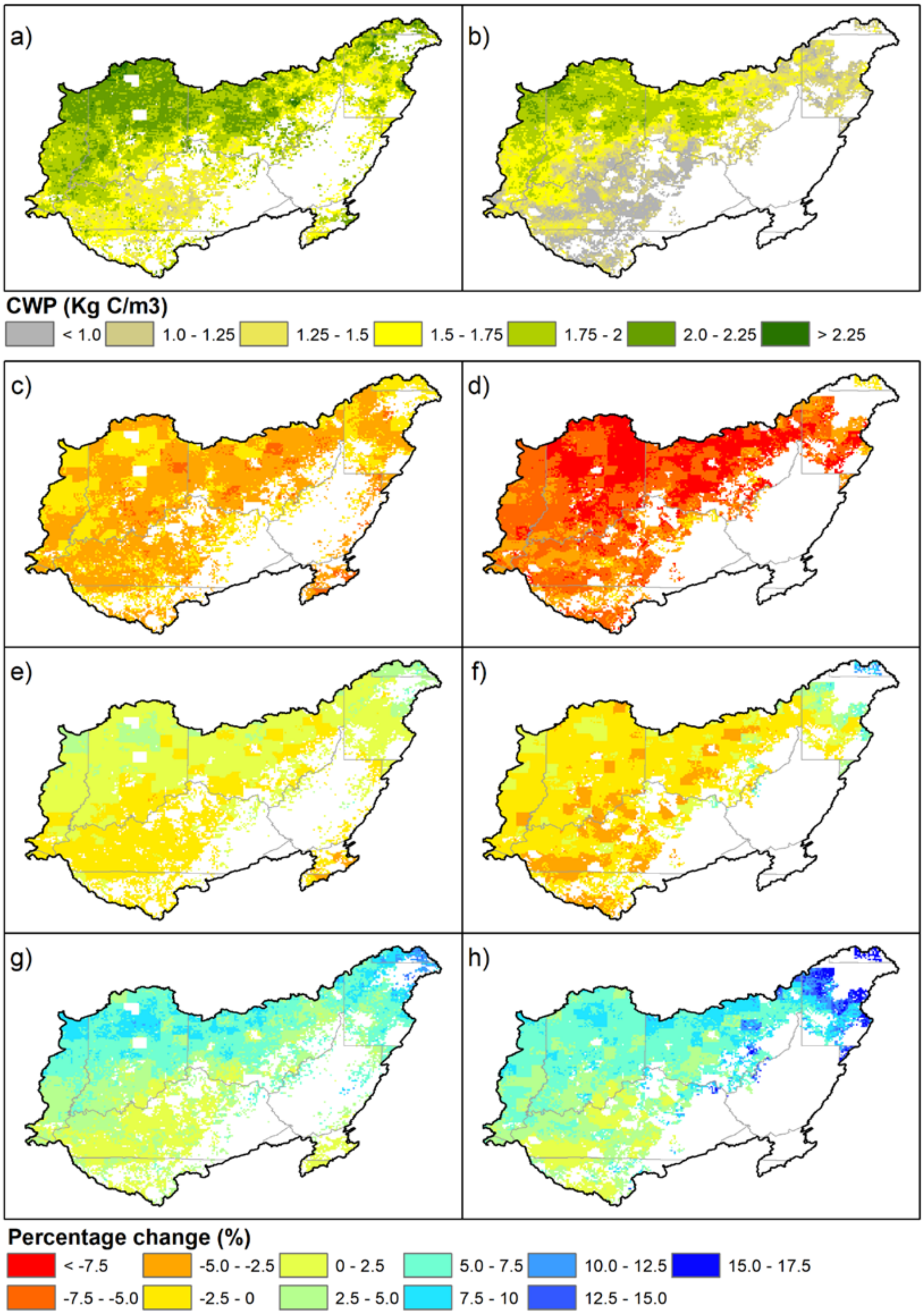


Figure 5.5 Spatial distribution of the mean annual (1979 - 2018) crop water productivity (CWP) in the ORB region (a, b), and the percentage change from the baseline CWP owing to CT (c, d), RT (e, h), and NT (g, h) (Left panel is for corn and right panel is for soybean)

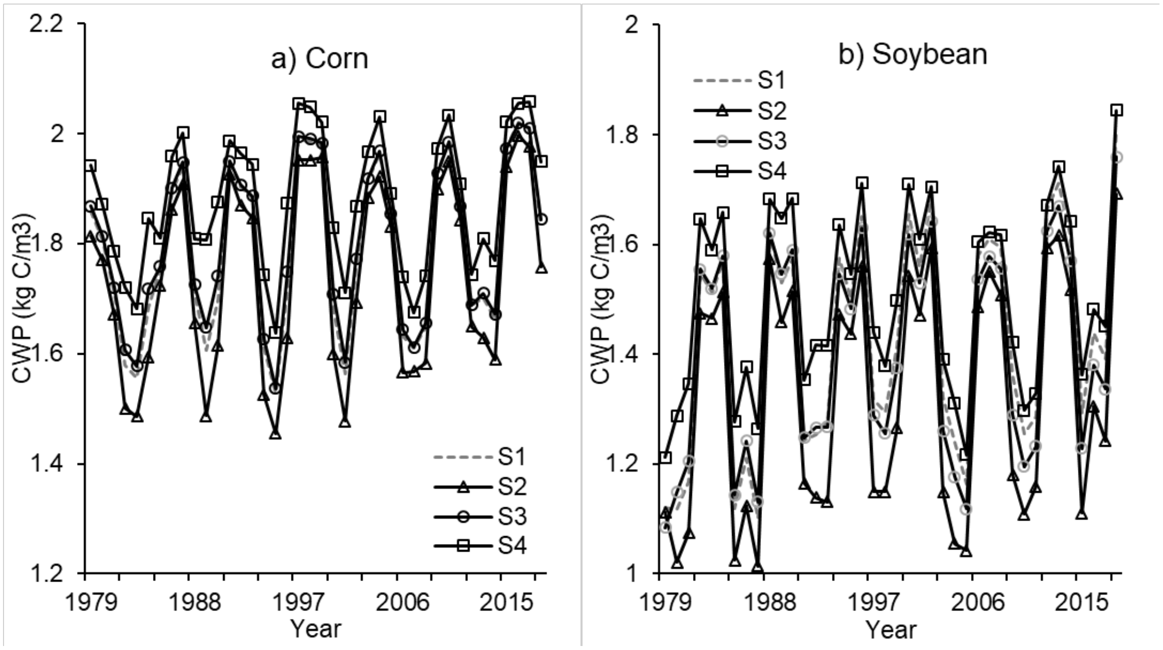


Figure 5.6 Temporal changes in crop water productivity (CWP) under different simulation experiments for corn (a) and soybean (b) over the ORB region. S1, S2, S3, and S4 are different simulation scenarios as shown in Table 5.1

## 5.5 Discussion

All forms of tillage alter soil characteristics in the affected zone, mostly in relation to soil water content (O'Brien and Daigh, 2019). Changes in climate and management practices, as well as their interactions, affect soil water dynamics, surface energy balance, and consequently, GPP, ET, and CWP. While it is crucial to investigate the impacts of different tillage systems on these parameters at local sites with the specific environment and management conditions, there is an urgent need to understand their regional responses considering the widespread adoption of conservation tillage during the past several decades. The present study addresses knowledge gaps by presenting average CWP, GPP, and ET values and their changes under different tillage scenarios among corn and soybean systems in a spatially explicit manner, using an agroecosystem modeling approach. To the authors' knowledge, this is the first study to compare long-term CWP among different tillage systems for corn and soybean at a regional scale.

### 5.5.1 Tillage Management and CWP Issues

The results of the present study showed that, on average, across the ORB region, different tillage regimes had indistinguishable effects on GPP for corn or soybean croplands (Figure 5.3). This is not surprising considering that the ORB is often "water-rich" (Figure 5.2d, Adler et al., 2003) with plentiful rainfall as well as numerous major rivers and impoundments. Alterations in soil water dynamics caused by different tillage methods would probably not limit the crop available water in the basin. Soil and water conservation technologies do not always lead to enhanced crop productivity (Hellin and Schrader, 2003). Previous studies suggested that dry areas where crop productivity is limited by soil water could potentially benefit from NT, but this is not the case for humid areas (Huang et al., 2018; Pittelkow et al. 2015). Kumar et al. (2012) found that although NT and RT increased soil water capacity compared to CT in two long-term sites at eastern and northern Ohio, there was a slight trend with higher crop yield under conservation tillage than under CT on the well-drained soil but no significant difference among tillage systems at the poorly drained site. Climate and soil may be major factors influencing crop productivity response to tillage (Toliver et al., 2012). In southern Illinois, Kapusta et al. (1996) also observed no difference in corn yield among CT, NT, and RT on a silt loam soil after 20 years of each tillage treatment. Moreover, similar annual winter wheat GPP between CT and NT systems was recently reported in the inland Pacific Northwest region with a Mediterranean climate (Chi et al., 2016) and in the Southern Great Plains with a humid subtropical climate (Kandel et al., 2020) using the eddy covariance method.

In terms of ET, our results were consistent with current knowledge that conservation tillage decreases soil ET compared to CT in the study region (Figure 5.4). NT and RT decreasing ET by 7 ~ 12% and 4 ~ 6% relative to CT, respectively, in corn and soybean systems. This is an important finding in the ORB region, considering the definition of CWP. The enhancement in CWP found in NT and RT scenarios (Figure 5.5) was mainly due to the decrease in ET and minor change in GPP. In addition, our results showed that NT and RT reduced evaporation compared with CT (Figure 5.4). They did not alter transpiration, which also explained the negligible distinctions in GPP among different tillage scenarios. Surface residues create a barrier that reduces evaporation and increases infiltration (Irmak et al., 2019). As conservation tillage, especially NT, rendered more residue coverage on the soil surface than CT, less evaporation was allowed. Besides, tillage typically increases surface roughness, therefore, reducing albedo (Cierniewski et al., 2015) and increasing net radiation to the soil (Schwartz et al., 2010). However, the mechanisms of how different tillage types would affect surface albedo and the associated legacy effect on evaporation are still far from being clear. Our current results might underestimate or overestimate the decrease in evaporation due to conservation tillage. Soil water evaporation is generally not favorable to crop productivity, although evaporation does slightly cool the surface microenvironment (Klocke et al., 2009), altering the soil energy balance (O'Brien and Daigh, 2019). Thus, adopting conservation tillage can reduce non-beneficial water loss via evaporation and make the soil more productive by maintaining soil moisture. One concern that exists regarding residue cover in the conservation tillage systems is that it would retard spring seed germination because of slower soil warming (Blanco-Canqui and Lal, 2009) and subsequently may lead to reductions in crop productivity. For example, long-term tillage studies in Illinois (Kapusta et al., 1996) and Indiana (Griffith et al., 1988) reported less corn population in NT and RT systems than in CT systems. However, they suggested that population differences among tillage systems did not translate into a yield deduction when N fertilizer was applied.

The present study also showed that the difference in CWP between NT and CT scenarios was slightly higher in soybean systems (~ 14%) than that in corn systems (~ 8%, Figure 5.5). Tang et al. (2015) observed similar results using both eddy covariance measurement and MODIS products in Minnesota. The greater response of soybean CWP could be due to that soybean has a less water-efficient photosynthesis pathway than corn (C3 vs. C4, Dietzel et al., 2016), and enhanced soil water content due to NT and RT would benefit soybean water use more than corn water use.

### 5.5.2 Tillage Management's Role in the Carbon and Water Cycles under Climate Change

Increasing CWP under climate change will largely rely on management practices that will reduce soil water evaporation and shift the water use to more transpiration (Hatfield and Dold, 2019). Soil preparation plays a critical role in ensuring crop productivity and CWP in response to climate change. Our results support the theory that conservation tillage can make agroecosystem less susceptible to the negative impacts of climate change by partitioning more water into infiltration to maintain soil moisture, thus potentially reducing crop water stress during drought conditions. In addition, soils in the ORB are vulnerable to water erosion, particularly during heavy spring rains under CT systems. We found that compared to CT, NT, and RT decreased surface runoff but increased subsurface drainage in the study region (Figure 5.7). However, the sum of runoff and drainage did not vary among different tillage scenarios. This finding is consistent with that of Daryanto et al. (2017). The shift in water fluxes (i.e., ET, runoff, and drainage) among tillage systems further suggested the advantages of NT and RT in enhancing soil water storage. In addition, it is generally perceived that NT and RT can reduce soil carbon loss compare to CT, which will help maintain or build up soil carbon storage and improve soil structure for the long run, hence making the soil more sustainable. However, it should be noted that as NT and RT also increased subsurface drainage, they may potentially lead to more nutrient leaching. Daryanto et al. (2017) reported that leachate nitrate amount was greater under NT than under CT despite the similar nitrate concentration under both systems. Considering the plentiful rainfall amount with a high probability of an increasing trend in the ORB region, such a problem should be stressed in NT systems in terms of implementing necessary remedial measures. For example, water harvesting technologies (e.g., terrace farming and drainage ditches) can help further increase available water for crops and lower the risk of nutrient leaching (Liu et al., 2020).

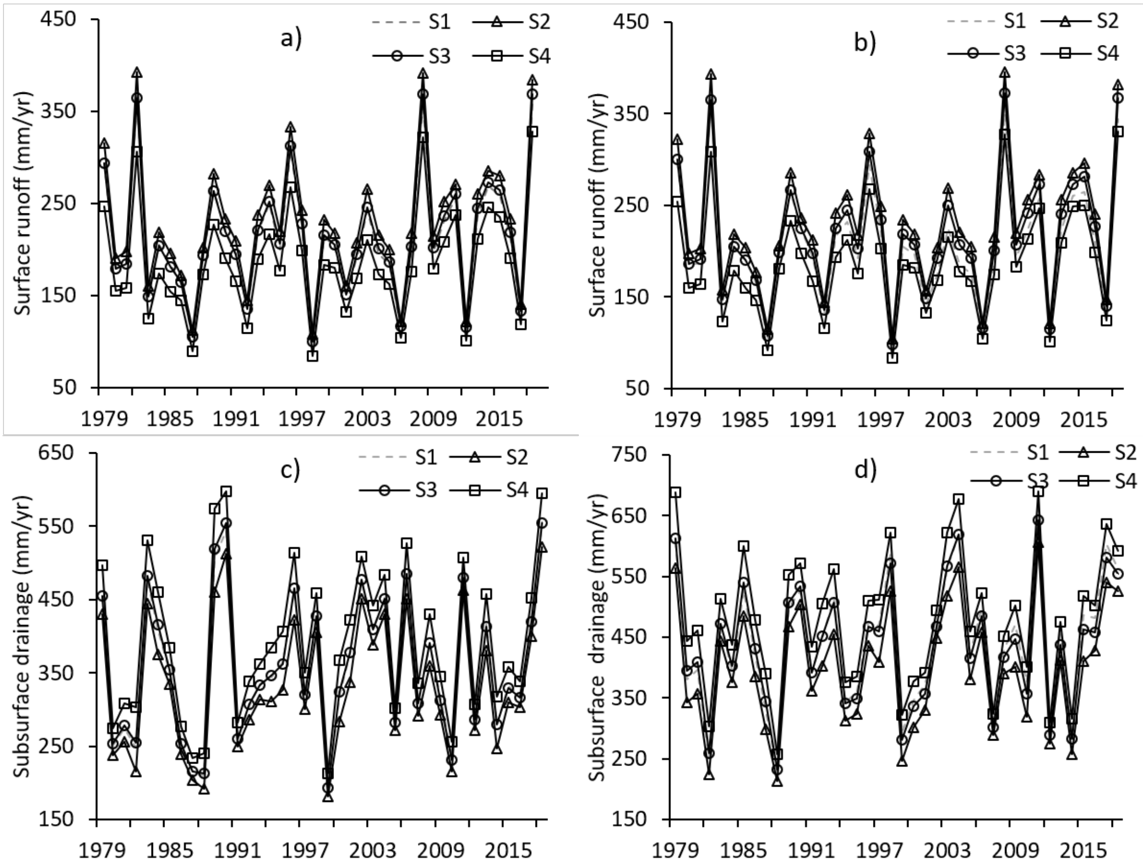


Figure 5.7 Temporal changes in surface runoff (a, b) and subsurface drainage (c, d) under different simulation experiments for corn (left panel) and soybean (right panel) over the ORB region (S1, S2, S3, and S4 are different simulation scenarios as shown in Table 5.1)

## 5.6 Conclusions

Process-based agroecosystem models provide an effective tool for quantifying CWP dynamics and underlying mechanisms as affected by different tillage management scenarios. To the best of our understanding, this study offered the first attempt to quantify tillage effects on crop-type-specific CWP at the regional scale. Model simulation results showed that if all the cropland in the ORB region were under the NT system, the corn and soybean CWP would increase by 2-7% and 2-9%, respectively. However, if all the cropland were under the CT system, the corn and soybean CWP would decrease ~3% and ~6%, respectively. Our results indicate that conservation tillage can be an effective management practice to enhance CWP in the ORB corn and soybean cropping systems, maintaining soil productivity. This benefit is mainly due to lower water loss through non-beneficial evaporation under conservation tillage systems. However, additional management practices, along with conservation tillage, are needed to control nitrogen loss. Future research should give more attention to the synergic effects of conservation tillage and other management (e.g., nutrient and water management) on agroecosystems.

# CHAPTER 6. CONCLUDING REMARKS

## 6.1 Summary

In this dissertation research, conservation tillage effects on crop yield, SOC, soil GHG emissions, and crop water productivity have been investigated at multiple scales, from local site to regional level (state and river basin). Different analytical methods and techniques, including meta-analysis, process-based models, field observations, and data synthesis, were utilized to evaluate the effectiveness of conservation tillage in ensuring crop productivity, promoting soil health, and mitigating and adapting to climate change.

In general, climate conditions, soil texture classes, and the duration of conservation tillage are the main factors influencing conservation tillage effects on agronomic and environmental variables. Through meta-analysis, this dissertation found that, overall, NT reduced soil GHG emissions and increased crop yield in dry climates compared to CT. No-tillage can reduce soil $CH_4$ emissions by 15.5%, with a concomitant increase in soil $N_2O$ emissions of 10.4%. These effects tended to diminish with the long-term duration of NT. Soil $CO_2$ emissions can be significantly reduced, with a yield benefit, under NT in dry climates. However, in humid climates, NT tended to increase soil $N_2O$ emissions and reduce crop yield, suggesting careful consideration of NT adoption in humid regions. Although NT cannot mitigate all three GHG emissions simultaneously, there was some evidence of a reduction in overall GWP under NT treatments given specific conditions, which needs to be verified with further investigations.

This dissertation also found that, generally, conservation tillage practices can enhance SOC stocks by 5% compared to CT. Such enhancement in SOC can be found in the entire soil profile (deep to 1.2 m). The SOC increment due to conservation tillage was higher in arid/warm than in humid/cool areas. In addition, soil texture was a major factor affecting the effectiveness of conservation tillage in enhancing SOC stocks. Such enhancement of SOC stocks due to conservation tillage tended to be higher in fine-textured soils than in coarse-textured soils.

Although the benefits of conservation tillage were well observed, it was not always the case due to the highly different site-specific conditions. In this dissertation, case studies have been conducted to explore the conservation tillage effects at the site and regional scales by using a modeling approach. The DLEM-Ag with a newly developed tillage module was first calibrated and validated against field observations of crop yield, SOC, and GHG emissions at a long-term site in Lexington, Kentucky. The improved model was then applied to explore the effects of NT on soil carbon dynamics in a continuous maize cropping system. The simulated results suggested that NT, together with cover crops, would significantly enhance soil carbon sequestration in the context of climate change, gaining much more carbon benefits than those from NT alone. The factorial analysis further showed that elevated $CO_2$ and warming effects were the main contributors to soil carbon gains due to the promotion in cover crop growth. Finally, the model was applied to the regional level (i.e., Kentucky and the Ohio River Basin) for quantifying changes in carbon and water cycles in response to conservation tillage practices. The results suggested that NT can reduce GHG emissions in Kentucky and enhance crop water productivity in the Ohio River Basin.

This dissertation research demonstrates the integration of multiple quantitative approaches, such as meta-analyses and process-based simulations, for examining the comprehensive effects of conservation tillage. Results derived from this dissertation highlights the importance of fully quantifying the effectiveness of conservation tillage for food production and climate change adaptation and mitigation at multiple scales. More accurate predictions using the present approaches are dependent on more systematic data from additional tillage studies in the future and further development of the process-based models.

## 6.2 Outlook

This dissertation research mainly investigated the effectiveness of conservation tillage in enhancing crop production and mitigating GHG emissions at multiple scales. Although meta-analyses were carried out based on a large number of field studies, field experiments, especially long-term ones, are an invaluable tool in revealing the mechanisms and impact of management practices on agronomic and environmental variables. However, long-term measurements of soil GHG emissions are time- and resource-consuming, and consequently scarce in the literature. Also, long-term measures of GHG emissions and SOC can potentially involve artificial uncertainties, which may cause variation. Therefore, future research should involve well-designed experiments to discuss tillage effects on GHG emissions and SOC. A more systematic effort is needed to provide valuable data that can be used for data synthesis and meta-analyses.

Statistical methods are useful tools to test the relationship between environmental conditions (e.g., climate and soil properties) and target variables (e.g., crop yield, SOC, and GHG). They are essential supplements to field experiments that can save time, labor, and monetary investment. With more data available from field experiments, it is possible to develop more accurate statistical models to analyze and predict the performance of each practice in its field applications.

Process-based models provide a useful tool to analyze and predict the performance of management practices at multiple scales and to help establish management decisions. However, the accuracy of the simulation results depends not only on the embedded structure and processes in the model but also plentiful observation data. Field experiments and statistical methods contribute actual bases to the process-based model. They also provide valuable datasets for model calibration and validation. Therefore, future modeling studies should intimately be coupled with field observations to improve the representation of the management practices in models.

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**Huang, Y.**, W. Ren, L. Wang, D. Hui, J. Grove, X. Yang, B. Tao, and B. Goff (2018), Greenhouse gas emissions and crop yield in no-tillage systems: a meta-analysis. *Agriculture Ecosystems and Environment*, 268:144-153. https://doi.org/10.1016/j.agee.2018.09.002.

**Meetings and Presentations:**

**Huang, Y**., W. Ren, B. Tao, Y. Yang, and X. Zhu. 2019. Soil management for a sustainable future: simulating crop water productivity, soil water dynamics, and evapotranspiration in the conservation tillage system. *2019 ESA & USSEE Meeting*, Louisville, Kentucky. Aug 11- 16, 2019 (Oral Presentation)

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**Huang, Y**., W. Ren, B. Tao, X. Zhu. Nitrogen gas emissions and nitrate leaching dynamics under different tillage practices based on data synthesis and process-based modeling. *2017 AGU Fall Meeting*, New Orleans, Louisiana. Dec.11-15,2017 (Poster Presentation)

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