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Long-term no-till and stover retention each decrease the global warming potential of irrigated continuous corn

Running head: Global warming potential of irrigated corn

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ABSTRACT

Over the last 50 years, the most increase in cultivated land area globally has been due to a doubling of irrigated land. Long-term agronomic management impacts on soil organic carbon (SOC) stocks, soil greenhouse gas (GHG) emissions, and global warming potential (GWP) in irrigated systems, however, remain relatively unknown. Here, residue and tillage management effects were quantified by measuring soil nitrous oxide (N_2O) and methane (CH_4) fluxes and SOC changes (ΔSOC) at a long-term, irrigated continuous corn (*Zea mays* L.) system in eastern Nebraska, USA. Management treatments began in 2002, and measured treatments included no or high stover removal (0 or 6.8 Mg DM ha^{-1} yr^{-1}, respectively) under no-till (NT) or conventional disk tillage (CT) with full irrigation (n = 4). Soil N_2O and CH_4 fluxes were measured for five crop-years (2011 to 2015), and ΔSOC was determined on an equivalent-mass basis to ~30 cm soil depth. Both area- and yield-scaled soil N_2O emissions were greater with stover retention compared to removal and for CT compared to NT, with no interaction between stover and tillage practices. Methane comprised <1% of total emissions, with NT being CH_4-neutral and CT a CH_4 source. Surface SOC decreased with stover removal and with CT after 14 years of management. When ΔSOC, soil GHG emissions, and agronomic energy usage were used to calculate system GWP, all management systems were

net GHG sources. Conservation practices (NT, stover retention) each decreased system GWP compared to conventional practices (CT, stover removal), but pairing conservation practices conferred no additional mitigation benefit. Although cropping system, management equipment/timing/history, soil type, location, weather, and the depth to which ΔSOC is measured affects the GWP outcomes of irrigated systems at large, this long-term irrigated study provides valuable empirical evidence of how management decisions can impact soil GHG emissions and surface SOC stocks.

INTRODUCTION

The majority of global agricultural land base expansion during the last 50 yr has been due to increases in irrigated agriculture, which has contributed >40% of increased food production worldwide (FAO, 2011). In the U.S., intensification of agricultural systems through irrigation has provided about half the total value of all national crop sales on just 17 percent of the total U.S. agricultural land base (NASS, 2013; ERS, 2015). These high-yielding production systems, particularly irrigated corn (*Zea mays* L.), are targeted for crop residue removal to supply feedstock for second-generation bioenergy or livestock uses. Retention of corn residues (e.g., stover) is a recommended conservation practice to reduce soil erosion risk and support fertility by returning organic matter and nutrients to the soil (Wilhelm *et al.*, 2007; Johnson *et al.*, 2014; Karlen *et al.*, 2014). High levels of stover retained on the soil surface, however, can decrease grain yield by interfering with planting, decreasing germination, and increasing disease incidence (Verma *et al.*, 2005; Sindelar *et al.*, 2013). Partial removal of stover from less erosion-prone landscape areas, therefore, could be used to meet livestock or bioenergy feedstock demands while also maintaining or enhancing grain yield in warm temperate environments (Sindelar *et al.*, 2013; Karlen *et al.*, 2014; Wortmann *et al.*, 2016). Pairing stover removal practices with no-till (NT) is expected to further reduce potential risks of soil erosion and soil organic carbon (SOC) loss (Wilhelm *et al.*, 2007; Blanco-Canqui & Lal, 2009; Johnson *et al.*, 2014).

The magnitude and direction of SOC change and soil greenhouse gas (GHG) fluxes under irrigated management are variable, leading to uncertainty in global warming potential (GWP) predictions of stover and tillage practices. Higher grain yields under more intensive management can result in greater input-use efficiencies, thereby maintaining or even decreasing GWP relative to less intensively managed systems (Verma *et al.*, 2005; Snyder *et al.*, 2007; Venterea *et al.*, 2011; Grassini & Cassman, 2012). Further, irrigation-induced increases in grain yield and crop biomass relative to non- or deficit-irrigation (Norwood 1999; Maharjan *et al.*, 2014) has been proposed to increase SOC under irrigation, particularly in NT systems (Lal *et al.*, 1998). Recent studies, however, show that irrigated corn production systems have low potential to sequester C because high biomass production is offset by SOC losses from decomposition stimulated by irrigation, especially when the entire soil profile is considered (Verma *et al.*, 2005; Follett *et al.*, 2013; Schmer *et al.*, 2014a).

Although the potential for SOC storage is emphasized frequently as the major factor determining whether a production system will be a net GHG source or sink, SOC changes resulting from residue and tillage management practices could be offset or exacerbated by increased emissions of non-carbon dioxide (CO_2) GHGs such as nitrous oxide (N_2O) and/or methane (CH_4) (Li *et al.*, 2005; Grandy *et al.*, 2006; Mutegi *et al.*, 2010; Powlson *et al.*, 2011; Chen *et al.*, 2013; Schmer *et al.*, 2014a). Some researchers suggest that soil N_2O emissions are expected to be greater in NT compared to conventional tillage (CT) due to greater expected soil water-filled pore space (WFPS) and less aeration due to greater soil bulk densities (Linn & Doran, 1984; Baggs *et al.*, 2003; Rochette, 2008), but others predict smaller emissions under NT because of improved soil structure and cooler soil temperatures (Snyder *et al.*, 2009; Venterea *et al.*, 2011; Sainju *et al.*, 2012; Guzman *et al.*, 2015).

Similarly, crop residue management can result in contrasting responses, with residue removal decreasing (Jin *et al.,* 2014; Guzman *et al.*, 2015) or increasing (Congreaves *et al.*, 2016; Lehman & Osborne, 2016) direct GHG emissions from soils.

Because soil changes in response to management can take several years to detect, the ability to quantify management changes improves with experiment duration (Follett *et al.*, 2013; Lehtinen *et al.,* 2014; Jin *et al.*, 2015). Studies that empirically measure soil GHG emissions, SOC stock changes, and energy-inputs over a longer time frame (>3 yrs) are uncommon, and system GWP calculated for many studies rely on estimates or modeled values which are rarely verified for specific management systems in specific geographical locations. Here, we quantify the *in-situ* impacts of continuous crop residue removal and tillage on soil GHG emissions, SOC stocks, and resulting GWP in a long-term irrigated continuous corn system in eastern Nebraska.

MATERIALS AND METHODS

Site description

The study was located at the University of Nebraska-Lincoln's Agricultural Research and Development Center, Ithaca, NE (41°9'43.3" N, 96°24'41.4" W; 349 m asl). Soils are silt loams of the Tomek (fine, smectitic, mesic Pachic Argiudoll) and Filbert (fine, smectitic, mesic Vertic Argialboll) series, 0-2% slopes (Soil Survey Staff, 2014). The 30-year mean air temperature and precipitation are 9.8 °C and 74 cm, respectively (1981 to 2010) (NCDC; Station ID Mead 6 S). Historically, this area was cropped with corn (*Zea mays* L.), soybean [*Glycine max* (L.) Merr.], oat (*Avena sativa* L.), and alfalfa (*Medicago sativa* L.).

Continuous corn was established at the site in 2000. Details of the study design are reported in Schmer *et al.* (2014a). Briefly, the experiment is a randomized complete block design (n = 6 blocks) with factorial treatments in split plots. Tillage is the whole-plot factor (NT, CT; 9 m x 45.6 m) and stover removal rate is the subplot factor (none, 0%; medium, ~35%; high, >70%; 9 m x 15.2 m). All treatment combinations were in place by 2002. Nitrogen fertilizer was side-dressed at V4 to V6 stages at varying rates from 2001 to 2006. Since 2007, N fertilizer has been side-dressed as granular urea (46-0-0) at 202 kg N ha^{-1} yr^{-1} using a 12-row applicator with injector knives placing N between rows to a soil depth of 10 to 15 cm. The disk treatment area is tilled each year, usually just before planting in spring, to a depth of 15 to 20 cm. Irrigation was applied with a solid-set sprinkler until 2002, then with a linear-move irrigation system since that time. Irrigation amounts ranged from 25 to 266 mm from 2001 to 2015, with a 14-year average of 116 ± 17 mm. Corn stover is removed in the fall after grain harvest using a flail chopper set to a 10 cm cutting height.

Crop productivity

Plant production values, soil GHG emissions, and soil organic C changes are reported here for only the “none” and “high” stover removal levels under NT and CT in four of the six blocks where GHG emissions were measured (see below). Annual aboveground dry matter (DM) production was measured by hand from a 0.76 m x 3.04 m area shortly after physiological maturity (September or October) each year. Ears were removed, then stalks cut at ground level, chopped, weighed, and a subsample dried at 60 °C until constant mass. Ears were dried, weighed, and shelled to calculate DM grain yields. Cob weights were added to the DM stalk biomass to calculate total non-grain DM (i.e., stover) production. Harvest index (HI) was calculated as the ratio of grain to total aboveground biomass production.

Soil greenhouse gas emissions

Soil GHG emissions were sampled from four of six treatment blocks from April 2011 through May 2016 for a total of 100 sampling events that spanned five full crop-years (i.e., >1500 individual flux observations). In April 2011, four blocks were selected at random for intensive soil GHG measurement using static vented chambers. The same four blocks have been sampled over time as per the USDA-ARS standardized protocol for the Greenhouse gas Reduction through Agricultural Carbon Enhancement network (GRACEnet) (Hutchinson & Mosier, 1981; Parkin & Venterea, 2011). Vented gas sampling chambers were covered with reflective insulation, and both chambers and sampling bases that were installed in the field were constructed of 20 gage stainless steel. Each base covered an area of 1707 cm^2 (52.7 cm x 32.4 cm) and was installed to a soil depth of 5 to 7 cm, resulting in a base height of 5 to 7 cm aboveground. One chamber base was placed well within each treatment plot and oriented perpendicularly to rows such that the short edge was parallel to the row and the remainder of the base extended into the between-row area. The base footprint approximated a 1:2 ratio of (within-row)-to-(between-row) soil microsites. The between-row microsite included the injection furrow where urea fertilizer was applied. Bases were removed and re-installed throughout the growing season to accommodate field management operations (e.g., tillage, planting, fertilization, harvest). After any reinstallation, bases were allowed to equilibrate for at least 24 hr (but usually 2 to 4 d) before gas sampling to minimize immediate soil disturbance effects (Reicosky *et al.*, 2005).

At each sampling event, headspace gas samples were collected with syringes then injected into evacuated vials at four evenly spaced time-points over 30 min (0, 10, 20, 30 min). To account for diurnal variability and also approximate daily average temperature, sampling was conducted early to mid-morning on each sampling date. Sampling occurred every 5 to 10 d during the growing season (May to September), with greater frequency following tillage operations and N fertilizer application. Non-growing season measurements were collected monthly as weather and ground conditions allowed.

Ancillary measurements at each sampling date included air temperature, soil temperature at 15 cm, and soil moisture from 0-to-15 cm depth as measured by a hand-held time domain reflectometer (FieldScout TDR 300; Spectrum Technologies, Aurora, IL, USA) with a site-specific calibration. Volumetric soil water content was converted to water-filled pore space (WFPS) using soil bulk density values.

Sample GHG concentrations were measured within 10 d of collection using a headspace autosampler (CombiPAL; CTC Analytics, Zwingen, Switzerland) connected to a gas chromatograph (450-GC; Varian, Middelburg, The Netherlands) equipped with separate detectors for the simultaneous measurement of N_2O (electron capture detector), CH_4 (flame ionization detector) and CO_2 (thermal conductivity detector). Soil CO_2 data are not presented here. Soil N_2O and CH_4 fluxes were calculated as a linear or quadratic change in headspace gas concentration over time within the enclosed chamber volume (Wagner *et al.*, 1997; Venterea *et al.*, 2011), and soil N_2O fluxes were corrected for suppression of the surface-atmosphere concentration gradient (Venterea, 2010). Fluxes were considered non-zero if rates were greater than (i.e. production) or less than (i.e. consumption) the flux detection limits calculated on the basis of analytical precision, chamber deployment time, and ambient gas concentration (Parkin *et al.*, 2012). Area-scaled N_2O and CH_4 emissions were estimated by linear interpolation of flux rates between sampling dates, then summing daily rates over each growing season (i.e., trapezoidal integration method). Yield-scaled N_2O emissions, or greenhouse gas intensity (GHGI), were calculated by dividing area-scaled N_2O emissions by DM grain yield. Total non-CO_2 GHG emissions were calculated by summing area-based annual fluxes of N_2O and CH_4 after converting to CO_2 equivalents (Mg CO_2 eq ha^{-1}) using GWP values of 298 and 25, respectively (IPCC AR4, 2007).

Soil organic carbon change and system global warming potential

Soil samples for SOC and GWP calculations were collected in May 2001, November 2010, and November 2014. In 2001 and 2010, four cores (3.5-cm diameter) were sampled from each subplot and composited by depth increment (0-15, 15-30, 30-60, 60-90, 90-120, 120-150 cm) (Schmer *et al.*, 2014a). In 2014, two cores (4.1-cm diameter) were sampled from each subplot and composited by depth increment (0-7.5, 7.5-15, 15-30 cm). Soil bulk densities were determined for each soil depth increment using core volumes and dry weights (dried at 105 °C) for soils collected in all years. A subsample of fresh soil from each increment was passed through a 2-mm sieve, oven dried at 55°C (up to 48 h), ground to pass a 150-mm screen, and analyzed for total C and N concentrations (Follett & Pruessner, 2001). Surface soils were carbonate-free.

Due to differences in the time of year and depth increments sampled between years, changes in SOC stocks were calculated on an equivalent-mass basis that approximated a soil depth of 30 cm (Lee *et al.*, 2009). Changes in SOC (ΔSOC; Mg CO_2 eq ha^{-1}) were calculated for the same four experimental blocks measured for non-CO_2 GHG fluxes. Changes were calculated for three time periods: (1) 2001 to 2010 (early period); (2) 2010 to 2014 (later period); and (3) 2001 to 2014 (full period of study).

Global warming potential was calculated for each management system as total non-CO_2 soil GHG emissions plus total fuel usage from agronomic operations (Table S1) minus total SOC stock changes for each early and later time period (Mosier *et al.*, 2006; Cavigelli & Parkin, 2012). Total non-CO_2 GHG emissions for 2001 to 2010 were estimated for each management scenario (NT-None, NT-High, CT-None, CT-High) by multiplying the number of crop-years (n = 10) by the mean annual emissions measured from 2011 to 2015. Total non-CO_2 GHG emissions for 2010 to 2014 were calculated as the total emissions measured over those crop years. Total non-CO_2 GHG emissions for the full study period were calculated as the sum of the early and later periods. Fuel usage was estimated from other studies (Adler *et al.*, 2007; Schmer *et al.*, 2014b), converted to CO_2-equivalents per hectare using the most recent carbon intensity values for diesel (102.82 kg CO_2 eq GJ^{-1}) or electricity (105.62 kg CO_2 eq GJ^{-1}) (California Air Resources Board, 2014), and assumed to be constant across all years. A positive GWP value indicated that the system was a GHG source, and negative GWP value indicated that the system was a GHG sink. To standardize units for time, ΔSOC and GWP are presented on an annual basis (Mg CO_2 eq ha^{-1} yr^{-1}).

Statistical analyses

Soil temperature, soil WFPS, aboveground biomass production, grain yield, HI, stover removal, area-scaled GHG emissions (N_2O, CH_4, total non-CO_2 GHGs), and GHGI were analyzed using the GLIMMIX procedure of SAS (SAS Institute, 2014). Tillage (T), stover removal (R) and their interactions were considered fixed effects, with Year (Y) and block as random effects. For analyses including Year, the covariance structure for each response variable was selected based on the smallest Akaike Information Criterion (Littell *et al.*, 2006; Loughin, 2006). Five-year total emissions of non-CO_2 GHGs (2011 to 2016) and ΔSOC and GWP for each time period (2001 to 2010, 2010 to 2014, 2001 to 2014) were analyzed using the GLIMMIX procedure with T, R, and T x R considered as fixed effects and block as a random effect. All treatment effects were considered significant at $P < 0.05$. Data were ln-transformed for normality when necessary. Multiple comparisons between significant treatment responses were evaluated with Bonferroni-adjusted P-values using the SLICE option in the LSMEANS statement ($P < 0.05$). Values are reported as means ± standard error (se). All plot-level data is available on-line (USDA-ARS Data Portal, http://nrrc.ars.usda.gov/arsdataportal/) (Del Grosso *et al.*, 2013).

RESULTS

Environmental conditions

Average air temperature and rainfall conditions varied widely across the 2011 to 2015 crop years (Fig. 1a, b). Mean annual air temperature ranged from 9.1 to 11.4 °C, and the 5-year mean (10.5 °C) was 7 ± 5% above the preceeding 30-year average (9.8 °C). Mean annual precipitation inputs ranged from 43.2 to 82.2 cm, and the 5-year mean (63 cm) was 14 ± 10% below the preceeding 30-year average (74 cm) primarily due to two drought years (2012, 2013). Growing season air temperatures ranged from 20.7 to 23.1 °C, and irrigation inputs ranged from 2.5 to 16.6 cm per year.

Measured soil temperatures (15-cm depth) were greater in CT (15.7 ± 0.3 °C) compared to NT (15.3 ± 0.3 °C) across all 100 sampling dates during 2011 to 2015 crop-years (P_T = 0.0334). Measured soil WFPS (0-15 cm depth) was less in CT (45.7 ± 0.3%) compared to NT (50.9 ± 0.3%) across all sampling dates, and the difference between CT and NT was greatest during the 2012 drought year (40.5% and 47.9%, respectively) (P_{Y*T} = 0.0010). There were no main or interaction effects of stover removal on either soil temperature or soil WFPS.

Crop responses

Total aboveground production (grain + non-grain biomass, Mg DM ha^{-1} yr^{-1}) and grain yield differed across the 2011 to 2015 crop-years (P_Y < 0.005), but were not affected by tillage or stover removal treatments (Tables 1, 2). Total aboveground production was least in 2015, greatest in 2013, and averaged 18.17 ± 0.74 Mg DM ha^{-1} yr^{-1} across the five crop-years (Table 2). Grain yields were numerically smallest in 2012, greatest in 2013, and averaged 9.85 ± 0.46 Mg DM ha^{-1} yr^{-1} across the five crop-years (Table 2). Harvest index (HI) was marginally greater when stover was removed (HI_{High} = 0.547) compared to no removal (HI_{None} = 0.536) (P_R = 0.0817) (Table 1). Harvest index values were NT > CT in 2011 and 2015, but NT < CT in the 2012 drought year (P_{Y*T} < 0.005) (Table 2). Values for HI ranged from 0.47 to 0.60 under NT, and from 0.48 to 0.61 under CT. The quantity of stover biomass removed was not affected by tillage but differed by year (P_Y < 0.0001; Table 1), ranging from 4.90 Mg DM ha^{-1} in 2014 to 8.84 Mg DM ha^{-1} in 2011. The mean stover biomass removed over the 5 crop-years was 6.87 ± 0.64 Mg DM ha^{-1} and was equivalent to 84% of non-grain aboveground biomass.

Soil N_2O fluxes and greenhouse gas intensity

Measured daily soil N_2O fluxes ranged from -1.3 to 519.6 g N_2O-N ha^{-1} d^{-1}, with a median value of 2.2 g N_2O-N ha^{-1} d^{-1} and mean value of 14.6 ± 1.0 g N_2O-N ha^{-1} d^{-1} over the 2011 to 2015 crop-years (n = 1573) (Fig. 1c; Fig. 2). Eighty-four percent of all observations were non-zero fluxes. The greatest daily N_2O fluxes occurred one to three weeks following fertilizer application (Fig. 1). Peak fluxes occurred when soil temperature >25 °C and soil WFPS >60% (Fig. 2). Smaller N_2O efflux events occurred in February to March, corresponding with spring thawing of frozen soils. Although soils generally emitted N_2O, 2% of all non-zero observations were negative fluxes, indicating soil N_2O consumption.

Area-scaled annual N_2O emissions ranged from 0.4 to 7.2 kg N_2O-N ha^{-1} yr^{-1} over all treatments, with a mean annual value of 3.2 ± 0.2 kg N_2O-N ha^{-1} yr^{-1} for the 2011 to 2015 crop-years (Fig. 3a, d). Annual N_2O emissions did not differ between years in NT (2.9 ± 0.1 kg N_2O-N ha^{-1} yr^{-1}), but emissions were 42% and 73% greater in CT than NT for the 2013 and 2014 crop-years, respectively (Fig. 3a) (P_{Y*T} = 0.0174). Annual N_2O emissions were less when stover was removed (2.8 ± 0.2 kg N_2O-N ha^{-1} yr^{-1}) than the no stover removed treatment (i.e., stover retained) (3.5 ± 0.2 kg N_2O-N ha^{-1} yr^{-1}) (P_R = 0.0311) (Fig. 3d).

Yield-scaled annual N_2O emissions (GHGI) ranged from 0.034 to 1.081 kg N_2O-N Mg^{-1} grain over all treatments, with a mean annual value of 0.333 ± 0.019 kg N_2O-N Mg^{-1} DM grain for the 2011 to 2015 crop-years (Fig. 4a, b). Over all crop years, GHGI with NT (0.299 ± 0.021 kg N_2O-N Mg^{-1} DM grain) was less than CT (0.363 ± 0.027 kg N_2O-N Mg^{-1} DM grain) ($P_T = 0.0368$), and GHGI was less when stover was removed (0.289 ± 0.022 kg N_2O-N Mg^{-1} DM grain) compared to retained (0.373 ± 0.020 kg N_2O-N Mg^{-1} DM grain) ($P_R = 0.0107$).

Soil CH_4 fluxes

Measured daily soil CH_4 fluxes ranged from -23.0 to 97.0 g CH_4-C ha^{-1} d^{-1}, with a median value of -0.3 g CH_4-C ha^{-1} d^{-1} and a mean value of 0.2 ± 0.1 g CH_4-C ha^{-1} d^{-1} over the 2011 to 2015 crop-years (n = 1539) (Fig. 1d). Eighty-four percent of all observations were non-zero fluxes. Of all non-zero CH_4 fluxes, 44% of observations indicated CH_4 consumption (i.e., negative flux). There were no clear seasonal patterns of CH_4 fluxes over the five crop-years.

Area-scaled annual CH_4 fluxes ranged from -0.67 to 1.28 kg CH_4-C ha^{-1} yr^{-1} over all treatments, averaging 0.19 ± 0.04 kg CH_4-C ha^{-1} yr^{-1} for the 2011 to 2015 crop-years (Fig. 3b, e). Annual CH_4 fluxes did not differ between years in NT (0.07 ± 0.06 kg CH_4-C ha^{-1} yr^{-1}) and were not different from zero-flux in 4 of 5 crop-years. In contrast, CT soils emitted CH_4 in 3 of 5 crop-years, and annual CH_4 emissions were 399%, 89%, and 261% greater than NT soils in 2012, 2013, and 2015, respectively ($P_{Y*T} = 0.0024$) (Fig. 3b). There was no stover removal effect in CT (0.31 ± 0.13 kg CH_4-C ha^{-1} yr^{-1}), but NT soils emitted CH_4 in the high stover removal treatment (0.17 ± 0.06 kg CH_4-C ha^{-1} yr^{-1}) and were CH_4-neutral when no stover was removed (-0.03 ± 0.08 kg CH_4-C ha^{-1} yr^{-1}) ($P_{T*R} = 0.0161$) (Fig. 3e).

Total emissions of non-CO_2 gases

Area-scaled, total annual emissions of non-CO_2 GHGs (N_2O plus CH_4; Mg CO_2 eq ha^{-1} yr^{-1}) was dominated by N_2O (>99%) (Figure 3c, f). The mean annual GHG emissions for all treatments over the 5 crop-years was 3.0 ± 0.2 Mg CO_2 eq ha^{-1} yr^{-1}. The majority of emissions occurred during the growing season (81% to 93%). Total emissions did not differ between years in NT (2.7 ± 0.1 Mg CO_2 eq ha^{-1} yr^{-1}), but CT > NT for 2013 and 2014 crop-years ($P_{Y*T} = 0.0100$). Total emissions under high stover removal (2.6 ± 0.2 Mg CO_2 eq ha^{-1} yr^{-1}) were less than no removal (3.3 ± 0.2 Mg CO_2 eq ha^{-1} yr^{-1}) ($P_R = 0.0325$). There were no other treatment interaction effects for year, tillage, and stover removal on total annual non-CO_2 GHG emissions. The 5-year cumulative emissions did not differ between CT or NT, but was 23% less in the high stover removal treatment (12.2 ± 1.5 Mg CO_2 eq ha^{-1}) compared to no removal (15.8 ± 1.4 Mg CO_2 eq ha^{-1}) ($P_R = 0.0513$). Values for the mean annual total emissions (Mg CO_2 eq ha^{-1} yr^{-1}) used to estimate GWP for the early study period (2001-2010) were 2.8, 2.1, 3.5, and 2.8 for NT-None, NT-High, CT-None, and CT-High treatments, respectively.

Soil organic carbon changes

Soil organic carbon stocks (ΔSOC) in surface soils (fixed mass to ~30-cm soil depth) decreased over time in all management systems. Relative to 2001 SOC values, 2014 SOC changes were not different than zero for NT-None, marginal SOC loss under CT-None (-5%; $P = 0.1327$), and significant SOC losses under both NT-High (-9%) and CT-High (-10%) ($P < 0.01$). For all management periods, however, there were no interaction effects of stover and tillage management (Fig. 5, S1). Conservation practices of stover retention and NT resulted in no SOC change, but 9% of SOC was lost with a high level of stover removal and 7% of SOC was lost with CT over the 14-year study period (Fig. S1).

During the early study period (2001 to 2010), annual ΔSOC was affected only by stover management (P_R = 0.0100), with SOC loss occurring with stover removal (-1.4 ± 0.2 Mg CO_2 eq ha^{-1} yr^{-1}) (Fig. 5c). Although there were no tillage differences for annual ΔSOC, SOC loss rates were less than zero for both NT (-0.6 ± 0.4 Mg CO_2 eq ha^{-1} yr^{-1}) and CT (-1.0 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}) (Fig. 5a). During the later study period (2010 to 2014), there were no main or interaction effects of tillage or stover management on annual ΔSOC, and although values were not different than zero, all losses were numerically greater than annual ΔSOC measured in the early period (2001 to 2010). Over the full study period (2001 to 2014), annual ΔSOC was affected by stover removal (P_R = 0.0062) with annual SOC losses occurring only when stover was removed (-1.6 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}). Although there were no tillage differences over the full study period, annual ΔSOC was less than zero in CT only (-1.2 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}), indicating SOC loss.

Global warming potential

Annual GWP values were greater than zero for all management systems for all time periods, indicating that all systems were net GHG sources (Fig. 5b, d). During the early study period (2001 to 2010), annual GWP was affected by both tillage (P_T = 0.0157) and stover removal (P_R = 0.0299). Global warming potential was greater in CT (4.9 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}) compared to NT (3.8 ± 0.4 Mg CO_2 eq ha^{-1} yr^{-1}), and when stover was removed (4.9 ± 0.2 Mg CO_2 eq ha^{-1} yr^{-1}) compared to retained (3.9 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}). During the later period of the study (2010 to 2014), annual GWP was not affected by either tillage or stover removal, but were numerically greater in all systems compared to the earlier study period (mean GWP for all systems 5.2 ± 0.9 Mg CO_2 eq ha^{-1} yr^{-1}). Over the full study period (2001 to 2014), annual GWP was affected by stover removal (P_R = 0.0309) and, to a lesser extent, tillage ($P_T = 0.0817$). Global warming potential was greater when stover was removed (5.1 ± 0.4 Mg CO_2 eq ha^{-1} yr^{-1}) compared to retained (4.1 ± 0.4 Mg CO_2 eq ha^{-1} yr^{-1}), and in CT (5.2 ± 0.3 Mg CO_2 eq ha^{-1} yr^{-1}) compared to NT (4.0 ± 0.4 Mg CO_2 eq ha^{-1} yr^{-1}).

DISCUSSION

Although crop yields did not differ between management treatments in the last five years of this 14-year irrigated continuous corn study, non-CO_2 GHG emissions, surface SOC changes, and system GWP were affected by the main effects of year, tillage, and/or stover removal. Changes in SOC affected GWP more so than changes in soil non-CO_2 GHG emissions as a result of management. Management differences in SOC changes and soil GHG emissions, however, showed high spatial and temporal variability, emphasizing the importance of long-term measurements in capturing soil responses to management. Our findings suggest that stover management decisions in irrigated systems have more impact on SOC stocks than tillage management, but that both tillage and stover management decisions affect overall system GWP. Although the absence of management interactions indicated no added benefit in pairing conservation management practices (e.g., NT, stover retention) compared to using each practice individually, other agroecosystem goals such as benefits to soil health and reduced soil erosion risks will also affect producer decisions (Blanco-Canqui & Lal, 2009; Karlen et al., 2011; Stewart *et al.,* 2015).

Crop productivity

Neither tillage nor stover removal affected mean total aboveground biomass nor grain yields over the 2011 to 2015 crop-years. Stover removal, however, tended to increase HI (Tables 1, 2). In a previous study at this site, Schmer *et al.* (2014a) found greater grain yields under high stover removal (2001 to 2010) under NT but not CT. Other irrigated corn studies have also found improved grain yields with stover removal (Sims *et al.*, 1998; Halvorson and Stewart, 2015; Wortmann *et al.*, 2016). Warmer soil temperatures associated with stover removal likely provided more favorable conditions for corn germination, growth, and grain production (Kaspar *et al.* 1990; Sims *et al.*, 1998; Blanco-Canqui & Lal, 2007; Sindelar *et al.*, 2013; Halvorson & Stewart, 2015). In contrast, some irrigated corn studies have reported that stover removal did not affect yield (Biau *et al.*, 2013; Huang *et al.*, 2013; Kenney *et al.*, 2013), and progressive yield declines have been found in rainfed corn production sites when stover was removed (Wilhelm *et al.*, 2004; Blanco-Canqui & Lal, 2007).

While the present study found no tillage effects on production since 2010, lower grain yields with NT have been noted in other irrigated corn studies (Sims *et al.*, 1998; Halvorson *et al.*, 2006) as well as in rainfed studies (Grandy *et al.*, 2006; Bundy *et al.*, 2011; Ogle *et al.*, 2012; Sindelar *et al.*, 2015). In these studies, reductions in crop yield with NT were often attributed to cooler early season soil temperatures that potentially slowed early spring development and delayed tasseling. Although we found warmer soil temperatures in CT compared to NT in this study, temperature differences did not result in a CT yield advantage over the last five years.

Daily fluxes of soil N_2O and CH_4

For all management systems, daily N_2O emissions rose sharply one to three weeks after fertilizer application (Fig. 1). High emissions rates following N fertilizer application is expected, and the length of time to these high fluxes depends on N fertilizer type, application method, and soil conditions (e.g., moisture, temperature) (Halvorson *et al.*, 2008; Drury *et al.*, 2012; Venterea *et al.*, 2011; Decock, 2014; Maharjan *et al.*, 2014). Across all individual N_2O measurements (n = 1573), peak daily emissions occurred when soil temperatures were warmer than 25 °C and soil WFPS >60% (Fig. 2). Although there are many biotic and abiotic pathways that potentially contribute to soil N_2O emissions (Butterbach-Bahl *et al.*, 2013), the aerobic process of nitrification is generally considered the primary N_2O source when WFPS is 30-60%, and the anaerobic process of denitrification is the primary N_2O source when WFPS >60% (Linn & Doran 1984; Davidson, 1991; Bateman & Baggs, 2005). Depending on soil texture, many soils at 60% WFPS are at field capacity where soil oxidative (e.g. nitrification) and reductive (e.g. denitrification) processes co-occur while transitioning in response to soil oxygen levels (Linn & Doran 1984; Davidson, 1991). In this study, only 7% of all soil observations had WFPS >60%, and these soil moistures were associated with the greatest daily emissions rates measured over the 5 crop-years, especially when soils were warm. This suggests that episodic peak events during the growing season can heavily influence the total annual emissions of N_2O (Butterbach-Bahl *et al.*, 2013), emphasizing the importance of an adequate sampling frequency to accurately quantify annual N_2O emissions (Parkin 2008; Parkin & Venterea, 2010).

The optimum range for N_2O emissions is proposed to occur at 70-80% WFPS (Davidson, 1991; Chen *et al.*, 2013), beyond which N_2O levels are expected to drop as it undergoes complete stepwise denitrification to N_2 (Chapuis-Lardy *et al.*, 2007; Butterbach-Bahl *et al.*, 2013). In a review of European soils, Schaufler *et al.* (2010) found that N_2O emissions declined in very few of even the wettest soils, suggesting that moisture conditions beyond the optimum WFPS is infrequent in upland soils (Butterbach-Bahl *et al.*, 2013). Although we found that 2% of all non-zero N_2O fluxes were negative in this irrigated system study, there was similarly no evidence that N_2O consumption was occurring in the wettest soils. Rather, the majority of N_2O consumption events occurred in early spring (February, March), when median soil temperatures were cool (3.8 °C) and median soil moisture was 52% WFPS. During this time, a minor peak in soil N_2O effluxes also occurred, indicating that both oxidative and reductive processes were active when soils were undergoing freeze-thaw events (Fig. 1). Increases in N_2O effluxes in early spring associated with soil thawing have been measured in both rainfed (Congreaves *et al.*, 2016) and irrigated systems (Ellert & Janzen, 2008; Cui *et al.*, 2012). While the mechanisms for soil N_2O consumption are not yet clearly understood, increased N_2O consumption in cooler soils with more moderate soil moisture content could result from increased nitrifier denitrification, aerobic denitrification, or other shifts in active microbial community or abiotic processes (Chapuis-Lardy *et al.*, 2007; Ellert and Janzen *et al.*, 2008). Further, in colder climates where soils are frozen for extended periods, non-growing season emission events can constitute 30-90% of annual N_2O fluxes (van Bochove *et al.*, 2000; Teepe *et al.*, 2000; Congreaves *et al.*, 2016), but was only a minor contribution (~13%) to total annual emissions in this current study.

Upland agricultural soils are typically minor emitters or minor sinks for CH_4 (Bronson & Mosier, 1994; Mosier, 2006). Of the 1539 daily CH_4 measurements made in this study, 44% of all non-zero CH_4 fluxes were negative, indicating CH_4 consumption. Although CH_4 consumption is expected to occur in aerobic soils while CH_4 production occurs in anoxic soils (Conrad, 1996), we found no clear seasonal patterns between soil conditions and daily CH_4 fluxes over the five crop-years measured here.

Annual fluxes of non-CO_2 soil greenhouse gases

In this study, we found both area- and yield-scaled soil N_2O emissions were greater when stover was retained compared to removed and under CT compared to NT, with no interaction between stover and tillage practices. The absence of tillage and residue management interactions on soil GHG emissions has also been noted by others (Baggs *et al.*, 2003; Mahli & Lemke, 2007). Cumulative annual emissions of non-CO_2 soil GHGs differed between years for each tillage treatment such that emissions were greater under CT (Fig. 2, 3).

Area-scaled and yield-scaled (GHGI) annual N_2O emissions reported here were within range of measured or modeled values for other irrigated corn production systems (Adviento-Borbe *et al.*, 2007; Liu *et al.*, 2011; Cui *et al.*, 2012; Grassini & Cassman, 2012; Aguilera *et al.*, 2013; Huang *et al.*, 2013; Jin *et al.*, 2014; Zhang *et al.*, 2014). Further, both area-scaled and yield-scaled emissions showed similar responses to management treatments, which have also been noted in other irrigated and non-irrigated studies (Drury *et al.*, 2012; Huang *et al.*, 2013; van Kessel *et al.*, 2013). For area-scaled N_2O and CH_4 emissions, mean annual fluxes were relatively stable across the five crop-years under NT, but more variable under CT such that CT soils were stronger GHG emitters in some years (2013, 2014 for N_2O; 2012, 2013, 2015 for CH_4). Notably, NT was CH_4-neutral while CT was a CH_4 source. In addition, stover removal increased CH_4 emissions compared to no removal in NT but not CT. This contrasts with other irrigated corn studies that showed no tillage effects on soil CH_4 fluxes (Mosier *et al.*, 2006) or greater CH_4 emissions under NT compared to CT (Alluvione *et al.*, 2009). When converted to CO_2 equivalents, however, CH_4 contributed <1% of the total annual non-GHG emissions, thus the remainder the discussion focuses on N_2O below.

The variable effects of tillage on soil N_2O emissions emphasize how soils, weather, climate, and management duration can affect agroecosystem responses to management. A meta-analysis of 239 direct comparisons between NT/reduced tillage (RT) with CT found no overall tillage differences in soil N_2O emissions, but that in drier climates, NT/RT increased emissions in the short-term (<10 yr) by 56% but decreased emissions in the longer-term (>10 yr) by 27% compared to CT (van Kessel *et al.*, 2013). Similarly, a meta-analysis of only irrigated systems in Mediterranean climates also found no tillage differences (Aguilera *et al.*, 2013). Consistent with the larger van Kessel *et al.* (2013) analysis, however, Aguilera *et al.* (2013) also noted that NT/RT increased N_2O emissions by 72% compared to CT in experiments lasting <1 yr . In the present study, the 5-yr mean soil N_2O emissions after 10 yr of management was 22% lower in NT (2.3 ± 0.3 kg N_2O-N ha^{-1} yr^{-1}) compared to CT (3.3 ± 0.4 kg N_2O-N ha^{-1} yr^{-1}).

Soil N_2O emissions have been predicted to be greater in NT compared to CT due to higher bulk densities leading to wetter, less aerated soils (Linn & Doran, 1984; Baggs *et al.*, 2003; Rochette, 2008), but emissions have also been predicted to decrease in NT because of improved soil structure and cooler soil temperatures (Snyder *et al.*, 2009; Venterea *et al.*, 2011; Sainju *et al.*, 2012; Guzman *et al.*, 2015). Although CT soils were generally warmer and drier than NT in this study, we found that greater emissions resulted from more inter-annual variability in CT responses. For two of five crop years, soil N_2O emissions nearly doubled whereas emissions during the other three years were similar to NT. While it is not clear why emissions were high during those two years, the temporal variability observed in N_2O emissions between years as well as during the year are important factors to consider in accurately quantifying management effects on soil GHG fluxes. In addition, greater variability in weather, particularly the significant increase in mean annual air temperature and higher rainfall variability during the last 5 yr of this study, approximate climate changes that have been predicted for the central Great Plains region of the U.S. (Bathke *et al.*, 2014). Thus, management responses here may provide some insight into potential impacts of projected climate changes on intensively managed agroecosystems.

The high rate of stover removal used in this study decreased both area-scaled and yield-scaled annual soil N_2O emissions compared to no removal, likely due to a decrease in available C and N substrate as well as altered environmental conditions. Limited data on the effect of crop residue removal on soil N_2O emissions in warm temperate irrigated corn systems suggest that residue removal effects depend on residue quality (e.g., lignin, polyphenol, C:N ratio; Liu *et al.*, 2011), and that emissions are expected to decrease when residues are removed compared to retained (Heller *et al.*, 2010; Liu *et al.*, 2011; Huang *et al.*, 2013). Other rainfed studies in warmer climates have also noted lower soil N_2O emissions when crop residue is removed (Jin *et al.* 2014; Guzman *et al.*, 2015). In contrast, rainfed studies from colder climates have shown that stover removal can increase soil N_2O emissions (Congreaves *et al.*, 2016; Lehman & Osborne, 2016), possibly due to warmer soil temperatures or changes in moisture/aeration conditions that stimulate denitrifier activity. In a meta-analysis of 28 crop residue removal studies of mostly non-irrigated systems, Chen *et al.* (2013) found that soil N_2O emissions were positively correlated with the amount of residue returned to the soil, and that residue-derived emissions were comparable to those from N fertilizer addition. Trends observed by Chen *et al.* (2013) are consistent with IPCC estimates that 1% of both synthetic fertilizer N and crop residue N are lost to the atmosphere as N_2O (IPCC, 2007). Thus, residue removal would be expected to result in concomitant reductions in soil N_2O emissions.

Soil organic carbon changes

Losses of SOC occurred with stover removal and CT, indicating the importance of maintaining C input and reducing soil disturbance in maintaining SOC stocks in irrigated corn production systems. All soils lost C except for the no stover removal system, where SOC losses were not different than zero and were consistent with those previously measured on a fixed depth basis in Schmer *et al.* (2014a). Although the same SOC loss trends persisted and were somewhat greater in the later part of the study (2010 to 2014), no management effects were detected during this time. Greater SOC loss rates later in the study were unexpected, as rates of SOC change are predicted to decrease with time under long-term treatment (Stewart *et al.*, 2007). Change values for SOC, however, were highly variable during this period, perhaps due to variability in sampling and/or weather conditions. Overall, stover removal resulted in a 9% loss of SOC after 14 yr of management (2001 to 2014). This is consistent with previous reports from this site and a similar nearby irrigated no-till continuous corn site showing measured and modeled 10-yr SOC losses (2001 to 2010) in all management systems, with significant SOC losses in the surface 30 cm of soil (Liska *et al.*, 2014; Schmer *et al.*, 2014a; Wienhold *et al.*, 2015).

Stover removal in other irrigated corn systems have also resulted in surface soil C losses (Biau *et al.*, 2013; Kenney *et al.*, 2013; Halvorson & Stewart, 2015). Although increasing crop rotation complexity generally enhances SOC storage, limited SOC changes occur when continuous corn is converted to a corn-soybean rotation (West & Post, 2002). Thus, similar to SOC losses observed under continuous corn, residue removal in a rainfed corn-soybean system also resulted in SOC losses (Lehman & Osborne, 2016). Other corn rotational systems (e.g., corn-wheat (*Triticum aestivum* L.) have shown numerical but not statistically significant SOC losses after crop residue removal (Lemke et al., 2010; Huang et al., 2013). In some rainfed no-till corn studies, however, surface soils gained SOC over time regardless of stover removal level, with slower accrual when stover was removed (Follett *et al.*, 2012; Stewart *et al.*, 2015; Jin *et al.*, 2015). Tillage can interact with stover management such that stover incorporation also can lead to SOC increases, as found in a meta-analysis of European agricultural systems (Lehtinen *et al.*, 2014) and in a nearby irrigated study (Adviento-Borbe et al., 2007). In other tilled systems, no changes in SOC have been observed when stover was retained and incorporated (Blanco-Canqui and Lal, 2007; Bundy *et al.*, 2011; Powlson *et al.* 2011; Biau *et al.*, 2013). Under NT, however, stover removal can decrease SOC or slow accrual compared to no stover removal (Follett *et al.*, 2012; Sindelar *et al.*, 2014; Stewart *et al.* 2015). While there were no significant main or interaction effects with tillage in the present study, management systems ranked from greatest SOC loss to least SOC loss in the longer-term was CT-high > NT-high > CT-none > NT-none.

In a regional assessment of stover removal across the U.S. Corn Belt, Johnson *et al.* (2014) estimated that a minimum stover return rate of 5.74 ± 2.4 Mg ha^{-1} yr^{-1} was necessary to maintain SOC levels. In the present study, SOC losses occurred even when all stover (~8.3 ± 0.3 Mg ha^{-1} yr^{-1}) was returned to soil. Our results are consistent with expectations that irrigated systems have low potential to sequester C because high biomass production is offset by SOC losses from decomposition stimulated by irrigation (Verma *et al.*, 2005; Follett *et al.*, 2013; Schmer *et al.*, 2014a). Although surface (0- to 30- cm) SOC changes can occur, Schmer *et al.* (2014a) found that SOC stocks in the top 150 cm of the soil profile did not change with stover removal or tillage. While maintenance and accrual of SOC in surface soils are critical to overall soil health and fertility, further research is necessary in deeper soils to better understand how management affects terrestrial SOC dynamics and C storage potential of the full soil profile (Kravchenko & Robertson, 2011; Follett *et al.*, 2013; Schmer *et al.*, 2014a).

Global Warming Potential

Conservation practices (e.g., NT, stover retention) each decreased GWP compared to conventional practices (e.g., CT, stover removal) when ΔSOC values were combined with soil GHG emissions and estimated fuel and irrigation usage. Here, stover removal increased system GWP because SOC losses offset decreased soil non-CO_2 GHG emissions. An irrigated corn-wheat system in China similarly found that crop residue removal increased system GWP primarily due to the negative effect on SOC stocks (Huang *et al.*, 2013). In contrast, other researchers have found that continuous corn systems using NT have neutral or negative GWP, while CT systems are net GHG sources (Grandy *et al.*, 2006; Mosier *et al.*, 2006; Follett *et al.*, 2013). Regardless of management practice, all systems evaluated here were net GHG sources, similar to a nearby irrigated continuous corn production system which used eddy-covariance techniques to determine that it was also a net GHG source (Adviento-Borbe *et al.*, 2007).

Importantly, using NT and stover conservation practices together over the long-term (i.e., 14 years) did not confer any additional mitigation benefit in this irrigated continuous corn system. System GWP is expected to decrease over time as both rates of SOC change and soil GHG emissions adjust to management (Six *et al.*, 2004; Stewart *et al.*, 2007; Chen *et al.*, 2013; Lehtinen *et al.*, 2014), but variability in cropping system, management equipment/timing/history, soil type, location, weather, and the depth to which ΔSOC is measured will affect the GWP outcomes of irrigated systems globally. At the regional scale, Nebraska has the highest percentage of U.S. irrigated acres (~15%; NASS, 2014); thus, this long-term irrigated study provides a valuable empirical dataset of management impacts on crop yields, soil GHG emissions, and surface SOC stocks in the western U.S. Corn Belt. Such long-term agronomic and soil outcomes can be applied by local producers to make evidence-based management decisions that promote soil health, reduce soil erosion risks, and ensure production sustainability.

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Figure Captions

Figure 1. (a) Daily air temperature (°C) with growing-season mean daily air temperature (GS_T); (b) water inputs (mm d^{-1}) from precipitation (gray bars) and irrigation (black bars) with annual precipitation total, growing season precipitation total (GS_{Precip}), and total irrigation inputs; and effects of tillage (NT, CT) and stover removal (none, high) on soil fluxes of (c) N_2O (g N ha^{-1} d^{-1}) and (d) CH_4 (g C ha^{-1} d^{-1}) for 2011 to 2015 crop years. Arrows in (c) indicate N fertilizer application dates for each year.

Figure 2. Effects of soil temperature (soil T, °C) and soil water-filled pore space (WFPS, %) on daily N_2O fluxes (g N ha^{-1} d^{-1}) for 2011 to 2015 crop years.

Figure 3. Effects of tillage (a-c) and stover removal (d-f) on soil fluxes of N_2O (kg N ha^{-1} yr^{-1}), CH_4 (kg C ha^{-1} yr^{-1}), and total non-CO_2 GHGs (Mg CO_2 eq ha^{-1} yr^{-1}) for 2011 to 2015 crop years. In (a-c), different letters indicate yearly differences; tillage differences by year

denoted by *P > 0.05, ***P > 0.005. In panels (d-f), asterisks indicate stover differences. In panels (b, e), (†) indicate fluxes not different than zero.

Figure 4. Mean greenhouse gas intensity (kg N_2O-N Mg^{-1} DM grain) for 2011 to 2015 crop years by (a) tillage and (b) stover removal rate. Asterisks indicate a treatment difference (*P > 0.05, **P > 0.01).

Figure 5. Effects of tillage (a, b) and stover removal (c, d) on annual soil organic carbon changes (ΔSOC) and net global warming potential (GWP) (Mg CO_2 eq ha^{-1} yr^{-1}) for 2001-2010, 2010-2014, and the full 2001-2014 time horizon. Negative ΔSOC values indicate SOC losses. Within each time period, treatment differences are denoted by *P > 0.05, **P > 0.01, and §P = 0.0817. In panels (a, c), daggers (†) indicate SOC changes that are different than zero.

Table 1. Significance of *F* values for fixed sources of variation for total aboveground production (non-grain + grain, grain yield, harvest index, and stover removed for 2011 to 2015 crop years.

Source of Variation	df	Total aboveground production	Grain yield	Harvest index	Stover removed
		Pr > *F*			
Year (Y)	4	**0.0040†**	**0.0017**	**<0.0001**	**<0.0001**
Tillage (T)	1	0.6914	0.6383	0.4977	0.5833
Removal (R)	1	0.9228	0.4278	0.0817	**<0.0001**
Y x T	4	0.2890	0.1422	**0.0049**	0.3139
Y x R	4	0.2898	0.4271	0.2349	**<0.0001**

T x R	1	0.0798	0.0977	0.3505	0.5833
Y x T x R	4	0.3900	0.2805	0.2192	0.3139

†Bold text indicates significant P values ($P < 0.05$).

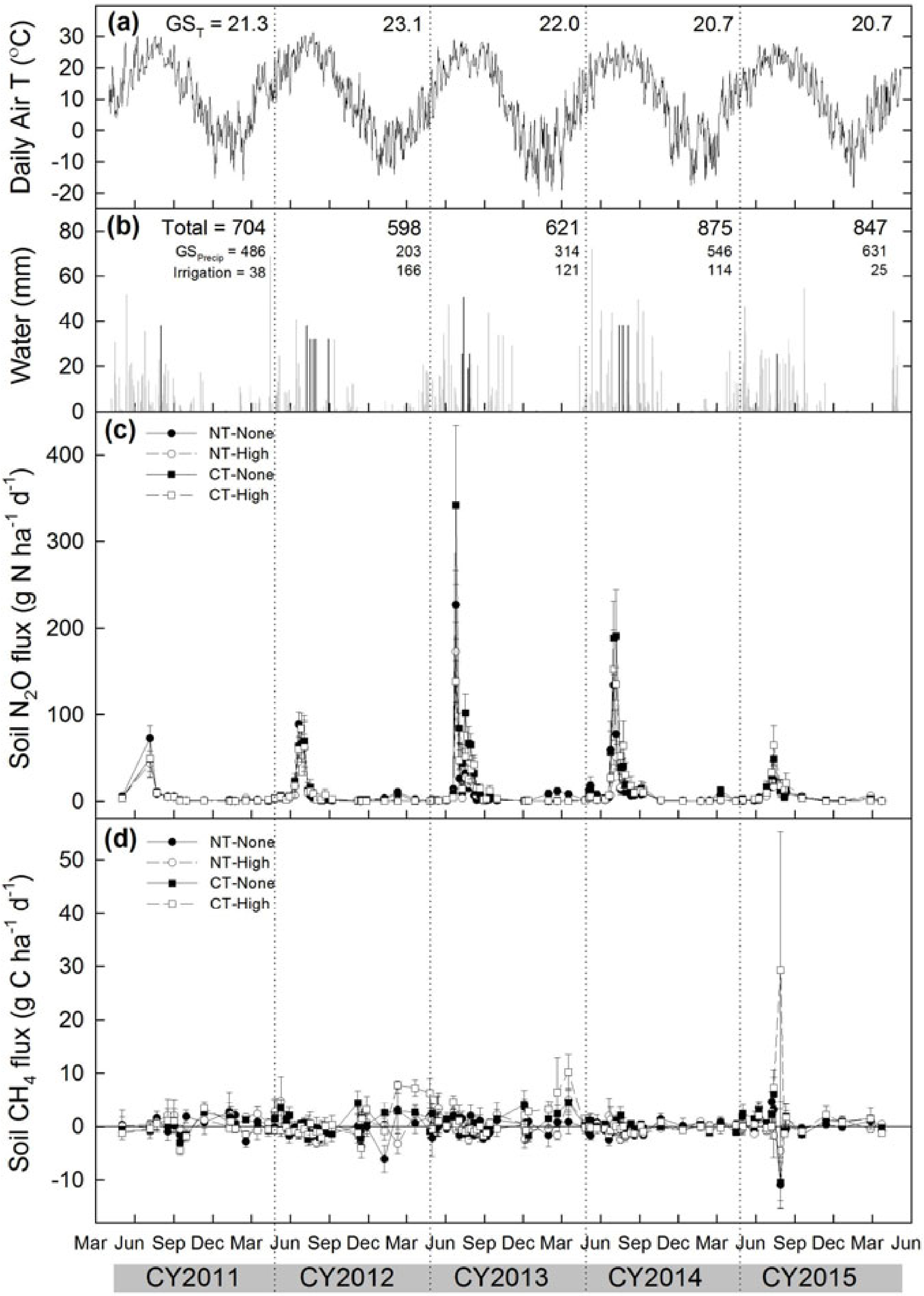
Table 2. Total aboveground production, grain yield, and harvest index in response to different soil management and stover removal practices for 2011 to 2015 crop years.

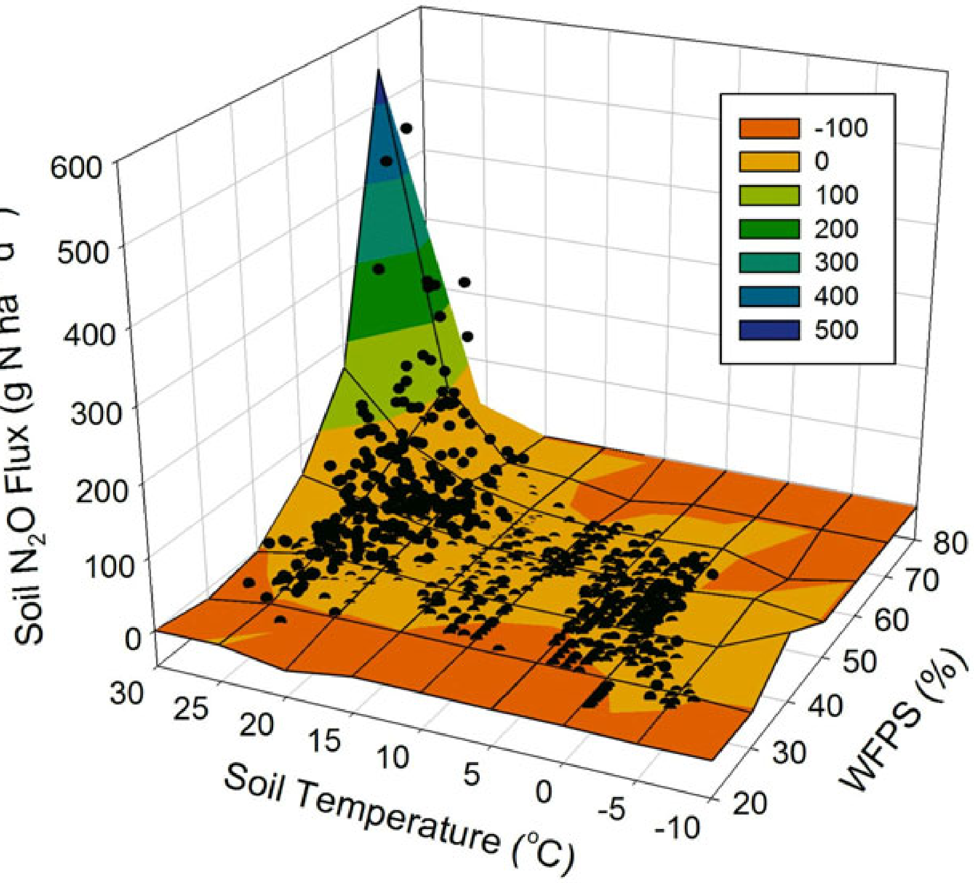
Removal level	No-till			Conventional tillage			Annual Mean
	None	High	Mean	None	High	Mean	
	Total aboveground production, Mg DM ha^{-1}						
2011	18.37	19.70	19.04	18.55	19.65	19.10	**19.07 b**†
2012	17.24	18.43	17.84	20.99	14.71	17.85	**17.84 ab**
2013	20.33	18.87	19.60	21.40	20.69	21.05	**20.32 b**
2014	16.57	18.63	17.60	17.26	18.29	17.77	**17.69 ab**
2015	16.56	18.58	17.57	14.15	14.33	14.24	**15.91 a**
Mean	17.81	18.84	18.33	18.47	17.53	18.00	18.17
	Grain yield, Mg DM ha^{-1}						
2011	9.54	10.12	9.83	8.67	9.45	9.06	**9.44 b**
2012	7.88	9.16	8.52	10.89	7.69	9.29	**8.91 ab**
2013	11.18	10.57	10.88	11.45	11.70	11.58	**11.23 b**
2014	9.38	11.72	10.54	10.49	11.05	10.77	**10.65 ab**
2015	9.81	10.41	10.11	7.97	7.92	7.94	**9.03 a**
Mean	9.55	10.40	9.98	9.90	9.56	9.73	9.85
	Harvest index‡						
2011	0.52	0.51	**0.52 AB**	0.47	0.48	**0.48 A**	0.50

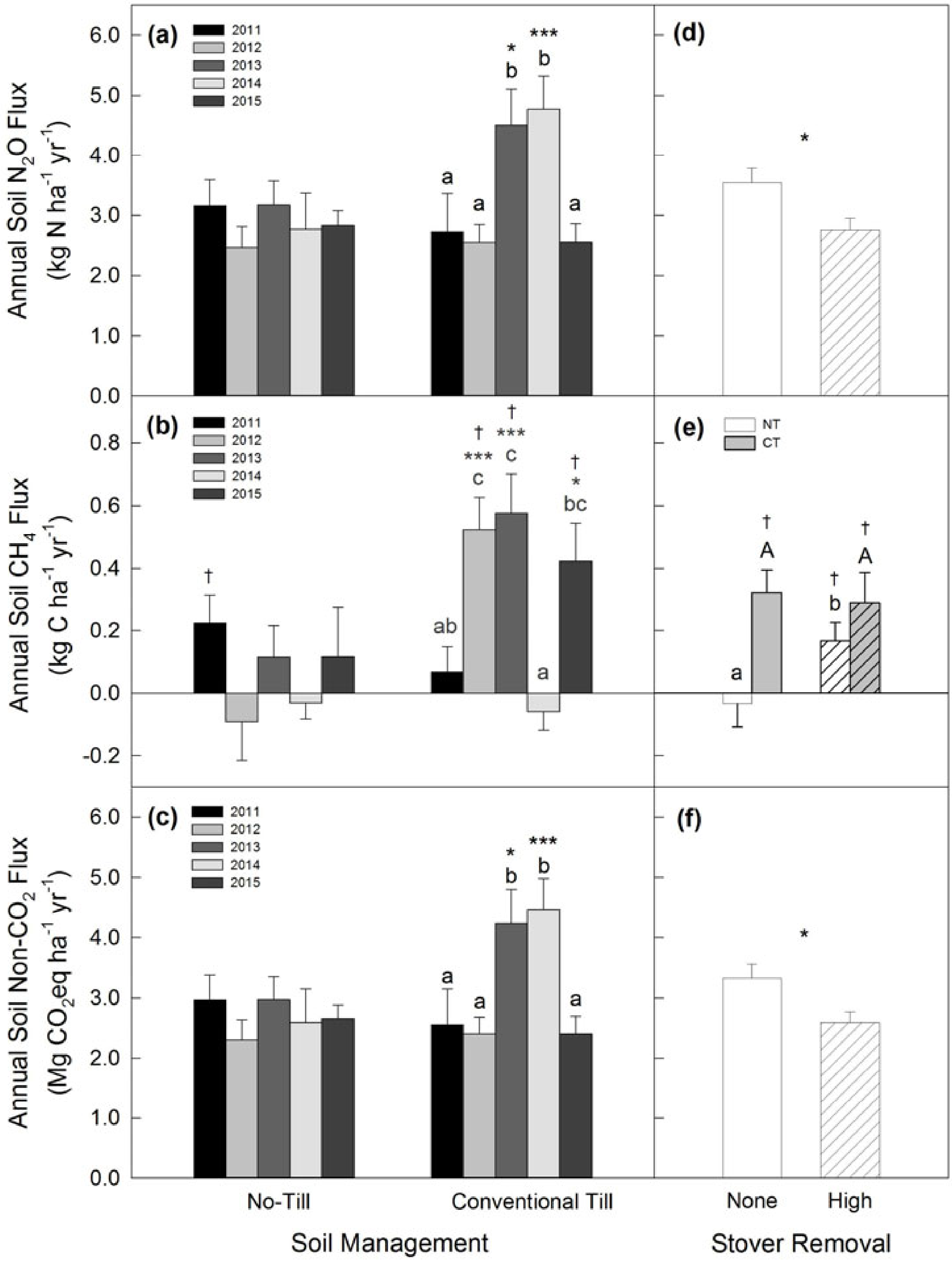
2012	0.45	0.50	**0.47 A***	0.52	0.52	**0.52 AB**	0.50
2013	0.55	0.56	**0.56 BC**	0.54	0.57	**0.55 BC**	0.55
2014	0.56	0.63	**0.60 C**	0.61	0.61	**0.61 D**	0.60
2015	0.59	0.56	**0.58 C***	0.55	0.54	**0.54 C**	0.56
Mean	0.54	0.55	0.54	0.54	0.54	0.54	0.54

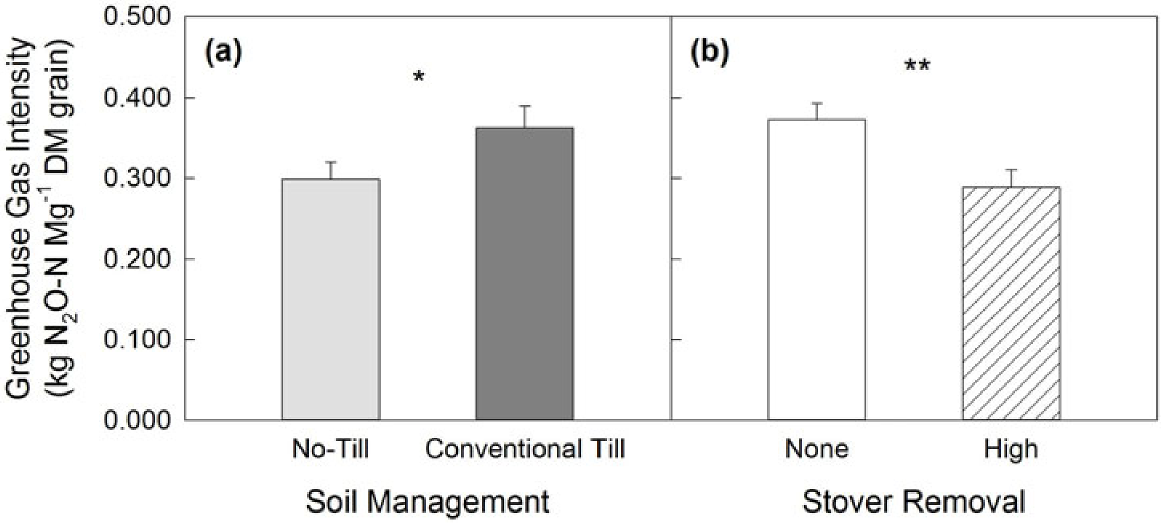
†Different lowercase letters indicate annual differences in aboveground production and yield ($P < 0.05$).

‡Different uppercase letters indicate annual differences within tillage treatments for HI, and asterisks indicate significant tillage difference within a given year ($P < 0.05$).

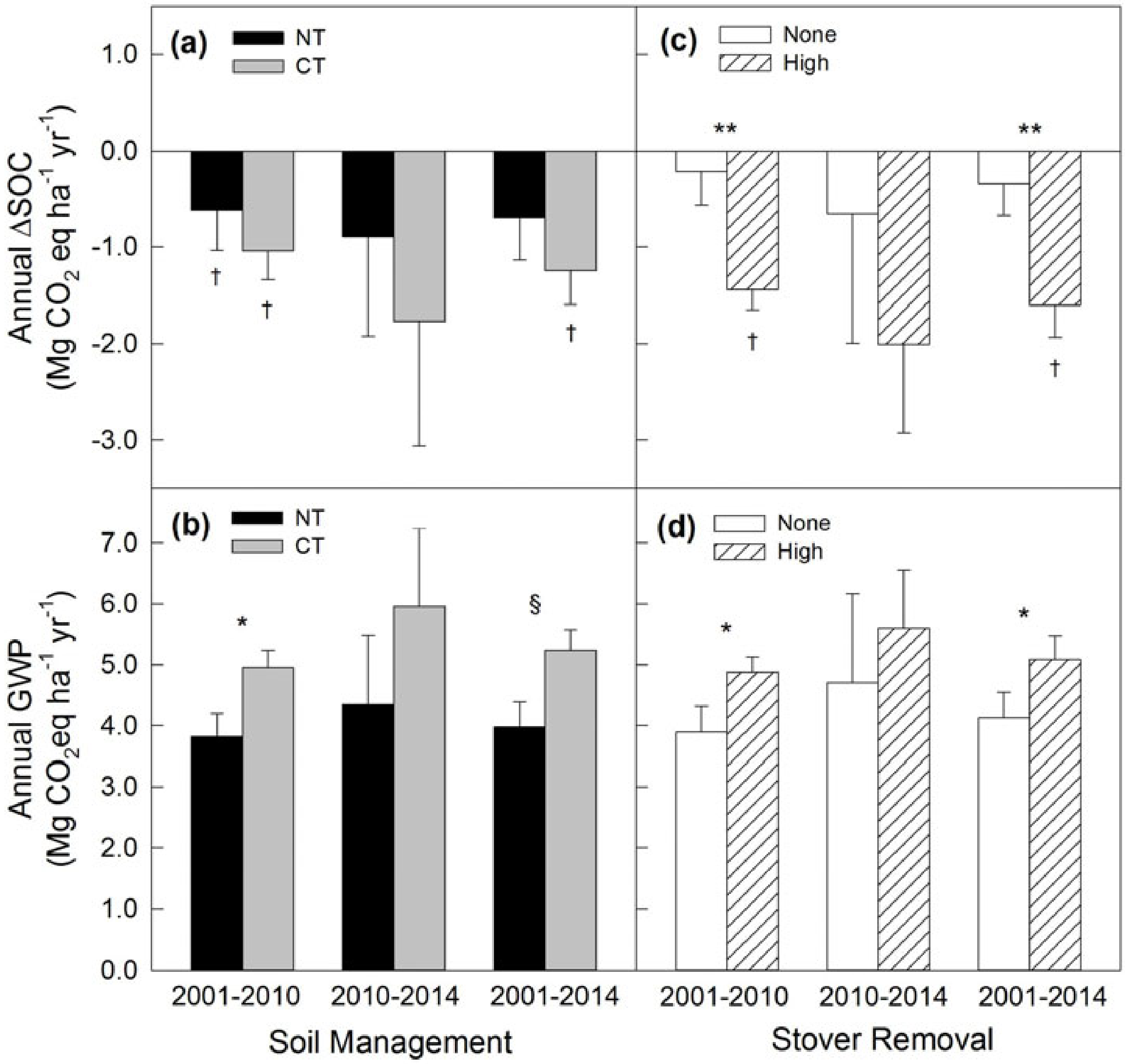

(a)
GS$_T$ = 21.3
23.1
22.0
20.7
20.7
Daily Air T (°C)
(b)
Total = 704
598
621
875
847
GS$_{Precip}$ = 486
203
314
546
631
Irrigation = 38
166
121
114
25
Water (mm)
(c)
NT-None
NT-High
CT-None
CT-High
Soil N$_2$O flux (g N ha^{-1} d^{-1})
(d)
NT-None
NT-High
CT-None
CT-High
Soil CH$_4$ flux (g C ha^{-1} d^{-1})
Mar Jun Sep Dec Mar Jun Sep Dec Mar Jun Sep Dec Mar Jun Sep Dec Mar Jun Sep Dec Mar Jun
CY2011
CY2012
CY2013
CY2014
CY2015


-100
0
100
200
300
400
500
600
500
400
300
200
100
0
Soil N_2O Flux (g N ha^{-1} d^{-1})
30
25
20
15
10
5
0
-5
-10
Soil Temperature (°C)
20
30
40
50
60
70
80
WFPS (%)


(a)
(b)
(c)
(d)
(e)
(f)
2011
2012
2013
2014
2015
NT
CT
Annual Soil N2O Flux (kg N ha-1 yr-1)
Annual Soil CH4 Flux (kg C ha-1 yr-1)
Annual Soil Non-CO2 Flux (Mg CO2eq ha-1 yr-1)
No-Till
Conventional Till
Soil Management
None
High
Stover Removal



Greenhouse Gas Intensity
(kg N2O-N Mg-1 DM grain)
0.500
0.400
0.300
0.200
0.100
0.000
(a)
*
(b)
**
No-Till
Conventional Till
None
High
Soil Management
Stover Removal



(a)
NT
CT
1.0
0.0
-1.0
-2.0
-3.0
Annual ΔSOC
(Mg CO_2 eq ha^{-1} yr^{-1})
†
†
†
(c)
None
High
**
**
†
†
(b)
NT
CT
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Annual GWP
(Mg CO_2eq ha^{-1} yr^{-1})
*
§
(d)
None
High
*
*
2001-2010
2010-2014
2001-2014
Soil Management
2001-2010
2010-2014
2001-2014
Stover Removal