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The greenhouse gas cost of agricultural intensification with groundwater irrigation in a Midwest U.S. row cropping system

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Abstract

Groundwater irrigation of cropland is expanding worldwide with poorly known implications for climate change. This study compares experimental measurements of the net global warming impact of a rainfed versus a groundwater-irrigated corn (maize)–soybean–wheat, no-till cropping system in the Midwest US, the region that produces the majority of U.S. corn and soybean. Irrigation significantly increased soil organic carbon (C) storage in the upper 25 cm, but not by enough to make up for the CO_2-equivalent (CO_2e) costs of fossil fuel power, soil emissions of nitrous oxide (N_2O), and degassing of supersaturated CO_2 and N_2O from the groundwater. A rainfed reference system had a net mitigating effect of −13.9 (±31) g CO_2e m^{-2} year^{-1}, but with irrigation at an average rate for the region, the irrigated system contributed to global warming with net greenhouse gas (GHG) emissions of 27.1 (±32) g CO_2e m^{-2} year^{-1}. Compared to the rainfed system, the irrigated system had 45% more GHG emissions and 7% more C sequestration. The irrigation-associated increase in soil N_2O and fossil fuel emissions contributed 18% and 9%, respectively, to the system's total emissions in an average irrigation year. Groundwater degassing of CO_2 and N_2O are missing components of previous assessments of the GHG cost of groundwater irrigation; together they were 4% of the irrigated system's total emissions. The irrigated system's net impact normalized by crop yield (GHG intensity) was +0.04 (±0.006) kg CO_2e kg^{-1} yield, close to that of the rainfed system, which was −0.03 (±0.002) kg CO_2e kg^{-1} yield. Thus, the increased crop yield resulting from irrigation can ameliorate overall GHG emissions if intensification by irrigation prevents land conversion emissions elsewhere, although the expansion of irrigation risks depletion of local water resources.

KEYWORDS
agriculture, carbon, corn, global change, greenhouse gas, groundwater, intensification, irrigation, maize, nitrous oxide, soybean, wheat

1 | INTRODUCTION

Global food security depends on irrigation to expand arable land area, intensify crop production, and provide a buffer from increasingly hot and dry growing seasons (Turral, Burke, & Faures, 2011). The recent expansion of irrigated area is expected to continue in coming decades in response to changing climate, growing populations, and increasing food consumption per capita (Konikow, 2011; Wada et al., 2010). Irrigation accounts for 90% of global consumptive water use, at least half of which is supplied by groundwater (Siebert et al., 2010), contributing to worldwide groundwater depletion (Famiglietti, 2014; Gleeson, Wada, Bierkens, & Beek, 2012) and, as a result, sea level rise by increasing the volume of global surface water (Konikow, 2011; Wada et al., 2016). Irrigation can increase the global warming impact of agriculture (Mosier, Halvorson, Peterson, Robertson, & Sherrod, 2005; Mosier, Halvorson, Reule, & Liu, 2006; Sainju, 2016; Sainju, Stevens, Caesar-TonThat, Liebig, & Wang, 2014; Trost et al., 2013, 2016), which is a major source of greenhouse gas (GHG) emissions to the atmosphere (IPCC, 2014). However, no existing studies of irrigated agriculture account for all potential GHG sources and sinks, nor represent irrigation use in the Midwest US (discussed below). The GHG impacts of groundwater irrigation are not included in the GHG inventory methods of the Intergovernmental Panel on Climate Change (IPCC) or the US Environmental Protection Agency (De Klein et al., 2006; USEPA, 2017a).

Irrigation has the potential to alter both GHG emissions and soil C sinks. Theory predicts and studies show that irrigation encourages soil microbial processes like denitrification, which produce nitrous oxide (N_2O; Hutchinson & Mosier, 1979; Panek, Matson, Ortiz-Monasterio, & Brooks, 2000; Robertson & Groffman, 2015; Trost et al., 2013)—a GHG with 298 times the global warming potential of carbon dioxide (CO_2) and a major driver of stratospheric ozone depletion (Myhre et al., 2013). Also, irrigation affects soil organic C (SOC) storage by increasing productivity and subsequent crop residue inputs but also by increasing decomposition (Lal, 2004).

Two more irrigation effects on GHG fluxes are specific to groundwater-fed systems. First, fossil fuel-derived energy is typically used to pump groundwater and drive sprinkler irrigation systems (West & Marland, 2002). Second, the partial pressures of GHGs dissolved in groundwater can be far greater than atmospheric equilibrium (i.e., supersaturated), resulting in the water degassing GHGs (i.e., GHG evasion) to the atmosphere during irrigation (Aufdenkampe et al., 2011; Turner et al., 2015; Wood & Hyndman, 2017).

Few studies have considered the implications of irrigation for GHG balances. Schlesinger (2000) explored the general concept of the global warming impact (GWI) of irrigation in drylands, accounting for CO_2 degassing from groundwater, increased SOC sequestration, fossil fuel emissions, and carbonate mineral precipitation. Mosier et al. (2005) and Mosier et al. (2006) calculated the GWI of irrigation in a Colorado corn–soybean system, accounting for soil GHG emissions, fossil fuels, nitrogen (N) fertilizer production and application, and SOC change. They found that fossil fuels used to pump the water were the major source of emissions, but no-till management with irrigation sequestered more CO_2 as SOC than the sum of CO_2-equivalent (CO_2e) GHG emissions. Neither of these studies, however, provided a rainfed control that would allow a calculation of irrigation impacts. Others (Jin et al., 2017; Sainju, 2016; Sainju et al., 2014; Trost et al., 2016) have assessed the GWIs of irrigated cropping systems and found that irrigation accounted for a major portion of emissions, sometimes offset by increases in SOC accrual, but none have included GHG degassing from groundwater.

Moreover, to date there have been no studies of the GWI of irrigation in the Midwest US, a major omission considering that the region produces about 60% of the corn and soybean in the US (USDA National Agricultural Statistics Service, 2014b). Twelve percent of annual U.S. corn and soybean production is irrigated, 25% of which is in the Midwest (USDA National Agricultural Statistics Service, 2014b). Irrigated area in the Midwest is expected to increase as summer rainfall declines and the number of dry days increases (Georgakakos et al., 2014; Pryor et al., 2014); irrigated area in Michigan alone increased by nearly 20% between 2007 and 2012 (USDA National Agricultural Statistics Service, 2014a).

Here we investigate how groundwater-fed irrigation affects the net GWI and GHG intensity of a Midwestern no-till cropping system by directly comparing irrigated and rainfed systems in a randomized complete block design. We include in our assessment groundwater degassing, changes in SOC, changes in soil N_2O emissions, and the CO_2 cost of energy used for pumping.

2 | MATERIALS AND METHODS

2.1 | Study site

We conducted this study at the Long Term Ecological Research (LTER) site at Michigan State University's Kellogg Biological Station (KBS), located in southwest Michigan (42°24′N, 85°24′W, elevation 288 m) within the northern U.S. Corn Belt. The site is located on a nearly level glacial outwash plain resulting from the Wisconsin ice sheet retreat ~18,000 years ago (Robertson & Hamilton, 2015). Soils are moderately fertile, well-drained loams (Typic Hapludalfs) developed on glacial outwash with intermixed loess (Crum & Collins, 1995; Luehmann et al., 2016). Crop yields at KBS and the surrounding county are similar to Midwest U.S. averages (Robertson et al., 2015).

Mean annual precipitation from 1987 to 2010 at KBS was 1,007 mm, about half of which fell as snow (NOAA, 2017). Precipitation from October through April exceeds evapotranspiration, and annual recharge is 230 mm (Hamilton, Hussain, Lowrie, Basso, & Robertson, 2018). Average growing season (May–September) precipitation was 467 mm (1987–2010); during the course of this study, there was a growing season drought (336 mm) in 2012, and the 2017 growing season was almost as dry (363 mm; Figure 1). Evapotranspiration is typically about 60% of annual precipitation (Hamilton et al., 2018). Mean summer and winter temperatures from 1987 to 2010 were 22°C and −2.6°C, respectively.

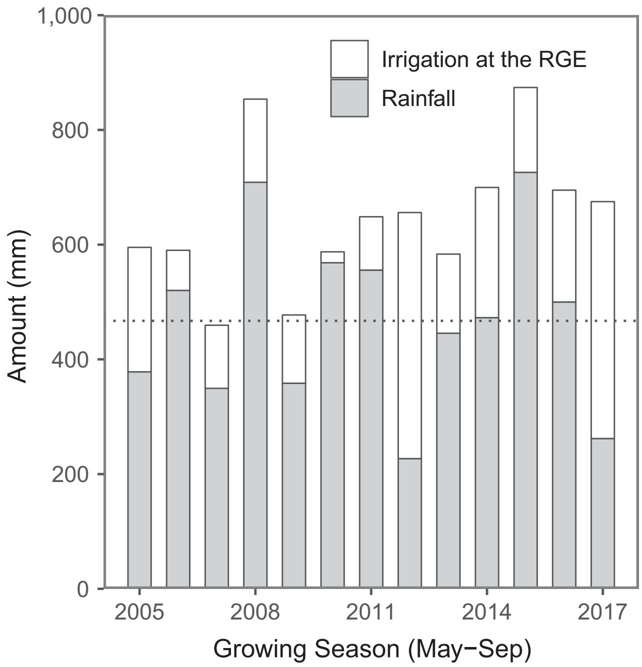

FIGURE 1 Growing season (May-September) rainfall and irrigation amounts at the Kellogg Biological Station's Long Term Ecological Research Resource Gradient Experiment (RGE). The dotted line is the mean growing season rainfall for 2005 to 2017 (467 mm). Labeled years are corn years, followed by soybean then wheat. Droughts occurred in 2012 and 2017

This study was conducted at the KBS LTER Resource Gradient Experiment (RGE) and draws on data from the adjacent KBS LTER Main Cropping System Experiment (MCSE; Robertson & Hamilton, 2015). The RGE is managed as a no-till corn (*Zea mays* L.)–soybean (*Glycine max* (L.) Merr.)–winter wheat (*Triticum aestivum* L.) rotation and includes irrigated and rainfed treatments. The RGE was established in 2005 and includes nine N fertilization treatments (F1-F9) replicated across eight blocks, half of which are irrigated with groundwater (irrigation details below). We used data from only one fertilizer treatment (F6) except as noted.

Nitrogen fertilizer was applied to the corn (168 kg N ha^{-1} year^{-1}) and wheat (112 kg N ha^{-1} year^{-1}) but not soybean, except in 2012. For corn and wheat years until 2013, liquid urea ammonium nitrate (UAN, 28-0-0) was broadcast with a sprayer. After 2014, UAN was knifed 13–15 cm below the soil surface. In 2012 only, as part of a different experiment, soybean was fertilized with UAN at 84 kg N/ha. F6 data were unavailable for SOC (more details below), so we used data from F5 (134 kg N ha^{-1} year^{-1} to corn and 90 kg N ha^{-1} year^{-1} to wheat). Both F5 and F6 N fertilization rates were similar to state averages: In 2016, the average application rate for corn grown in Michigan was 136 kg N/ha (USDA National Agricultural Statistics Service, 2017). We used the no-till treatment of the MCSE, managed identically to the rainfed RGE F6 treatment (Robertson & Hamilton, 2015), to supplement measurements from the rainfed RGE F6 treatment. Lime was applied to the RGE in 2012 with each plot receiving 0, 2.2, or 4.5 Mg dolomitic lime [$CaMg(CO_3)_2$] ha^{-1} (0, 1, or 2 ton/ac) based on soil tests. Rainfed plots and plots with higher N fertilization tended to receive more lime than the others.

From 2005 to 2011, irrigation was applied using solid set irrigation and was scheduled using the rainfall deficit. After 2011, irrigation was applied using a linear move irrigation system (Model 8000, 360° drops every 3 m, total length 70 m, Valley Irrigation, Valley, NE) and scheduled using a soil water budget. Total irrigation per season at KBS was similar to the average amount of irrigation applied to commercial corn fields in southwest Michigan, 0.18 m (7 in) as reported by Michigan Department of Agricultural and Rural Development (2015 Pers. Comm.) and southwest Michigan farmers (McGill, 2018).

2.2 | GHG degassing from groundwater

Two KBS groundwater wells used for irrigation were sampled for dissolved CO_2 and N_2O. These wells were sampled four times over the 2016 growing season, and the RGE well was also sampled ten times over the 2017 growing season. In the summer of 2016, additional wells were sampled from the surrounding southwest Michigan region: five within a 10 km radius of KBS (Kalamazoo River watershed) and 17 in the adjacent St. Joseph River watershed. In St. Joseph County, within the St. Joseph River watershed, 73% of corn cropland is irrigated (USDA National Agricultural Statistics Service, 2014a). At each dissolved gas sampling event water samples were also collected for analysis of major anions and cations. Water samples were filtered with a Supor 0.45 μm membrane filter (Pall Corporation; Ann Arbor, Michigan, USA) and refrigerated until analysis by membrane-suppression ion chromatography within one week (Dionex ICS-1100; ThermoFisher Scientific, Waltham, MA).

The dissolved gas sample collection method followed that of Hamilton and Ostrom (2007) and McGill (2018). At each well, samples were collected after the irrigation pump had been running for at least 15 min; often it had been running for an hour or more. We drew well water into a syringe without exposing the water to the atmosphere, an equal volume of ambient air was drawn into the syringe, the syringe was shaken gently for 5 min, and a gas sample was collected from the headspace. Gas samples were collected in triplicate at each site along with a fourth sample of ambient air.

Gas samples were analyzed at KBS within 30 days on a gas chromatograph (GC, Agilent 7890; Agilent Technologies, Santa Clara, CA) with a Gerstel MPS2XL automated headspace sampler (Gerstel, Mülheim an der Ruhr, Germany). CO_2 was measured with a Licor 820 Infrared Gas Analyzer and N_2O with a ^{63}Ni electron capture detector at 350°C. The GC system had a two-column back-flush configuration using Restek PP-Q 1/8″OD, 2.0 mm ID, 80/100 mesh, 3 m packed columns (Restek, Bellefonte, PA). The oven was set to 90°C. CH_4 concentrations were negligible and are not included here.

The dissolved CO_2 and N_2O concentration calculations followed those of Hamilton and Ostrom (2007) and McGill (2018) using the Bunsen solubility coefficient, Henry's Law, and the ideal gas law to calculate the gas concentration in the original water sample, accounting for the contribution from the original headspace (ambient air) gas concentrations (Weiss, 1974; Weiss & Price, 1980). The reported amount of gas that degasses from the groundwater upon equilibration with the atmosphere during irrigation was the difference between the atmospheric equilibrium concentration and the original water sample concentration, converted to g CO_2e m^{-2} $year^{-1}$ m^{-1} water applied (Schlesinger, 2000; Wood & Hyndman, 2017). N_2O was converted to CO_2 equivalents (CO_2e) by multiplying by a factor of 298—the global warming potential of N_2O relative to CO_2 over a 100-year period, as used by the IPCC in national GHG inventories (De Klein et al., 2006; Myhre et al., 2013).

Evapotranspiration of alkaline groundwater produces CO_2 when calcium carbonate ($CaCO_3$) precipitates and accumulates in arid soils (Schlesinger, 2000). In the humid Michigan climate, we do not observe $CaCO_3$ accumulation in irrigated soils, and therefore, we assume that any precipitation is balanced in the same year by redissolution with water infiltrating the soil, resulting in net neutral CO_2 emissions.

2.3 | Soil organic carbon

Initial SOC data were unavailable for the RGE, so we used a space-for-time substitution, where the difference between SOC in the irrigated and rainfed treatments in 2016 was used to estimate change in SOC due to irrigation over 12 years (sensu Syswerda, Corbin, Mokma, Kravchenko, & Robertson, 2011). We acknowledge that without SOC measurements from the start of the RGE, a difference between treatments could have resulted from one treatment either losing less or gaining more SOC than the other treatment (Olson, 2013). In November 2016, soil samples were collected using a double cylinder, 5.08 cm diameter hammer corer (AMS Soil Corer Model 404.05, American Falls, Idaho). We collected the 0–10 cm and 10–25 cm depths of each core separately. Deeper soils were not sampled because previous studies have shown that significant decadal SOC change at this site has occurred only in the upper soil profile (Kravchenko & Robertson, 2011; Syswerda et al., 2011). Three cores were collected per plot and analyzed separately. We sampled the two depths in F1 (unfertilized), F5 (~average N fertilization), and F8 (high N fertilization) plots from four blocks—two rainfed and two irrigated (n = 72 cores). Soils were sieved through a 4 mm mesh to remove roots and debris, then pulverized using a shatterbox (SPEX SamplePrep 8530 Enclosed Shatterbox, Metuchen, New Jersey), and analyzed with a CHN analyzer (Costech Elemental Combustion System CHNS-O ECS 4,010, Valencia, California). The CV for all replicate CHN samples was <10% (<5% in most cases). Soil C concentration was converted to kg C/m^2 using the cores' mean bulk density per depth stratum (the bulk densities among the cores were not significantly different). Because we did not have 2005 SOC data, rainfed SOC was subtracted from its irrigated N treatment counterpart, then divided by the 12 years of the experiment (2005–2016), and then by the total m of irrigation water applied over that period to provide kg SOC m^{-2} $year^{-1}$ m^{-1} irrigation water. We conducted regression analysis to determine the effect of irrigation on SOC.

2.4 | Soil N_2O emissions

Most of the measurements in this study were collected in 2016; however, the soil N_2O emissions were measured in 2013. They were collected for a different study at the RGE, and 2013 was the only period with simultaneous measurements of rainfed and irrigated soil N_2O emissions. Our final soil N_2O GWI calculation (details below) incorporates the 20-year average of annual rainfed soil N_2O emissions measured at the KBS LTER (Gelfand, Shcherbak, Millar, Kravchenko, & Robertson, 2016). Rainfall during the 2013 season was close to the 2005–2017 mean (Figure 1), and the 2013 air temperature maxima and minima were near the 2005–2017 means (Supporting information Figure S1). From May 8 to September 6, 2013, soil N_2O emissions were measured simultaneously in one rainfed and one irrigated block for N treatments F1, F3-F6, and F8 (n = 10 chambers). N fertilizer was applied on May 13, 2013, and wheat was harvested on 19 July 2013. Between those dates, 138 mm of irrigation water was applied. N_2O emissions were measured with an automated chamber system similar to Rowlings, Grace, Kiese, and Weier (2012) and described in detail in McGill (2018). Briefly, chambers were connected to an automated sampling system with a trailer-mounted GC. Each 0.5 × 0.5 m stainless steel chamber base was inserted 10 cm into the soil so that the base was flush with the soil surface. The chamber (0.5 × 0.5 × 0.15 m), fitted with an automated pneumatic lid, was attached to the base. Headspace gas was pumped to the GC (SRI Instruments, SRI 8610C, Torrance, CA, US) fitted with a ^{63}Ni electron capture detector for N_2O analysis (stainless steel column with Hayesep N: 2 m, 1/8 inch, 60/80 mesh, Alltech, US) in an oven at 60°C delivering gas at 30 ml/min to a detector at 330°C, with N_2 5.0 UHP carrier gas (Linde, US). The system collected four gas samples per 48-min incubation with four incubations per chamber every 24 hr. The GC analyzed three N_2O standards at the beginning and end of each full cycle and a N_2O check standard between each individual closure period. During rain and irrigation events, incubations were automatically aborted and chambers opened.

Hourly flux rates of N_2O (mg N_2O–N m^{-2} hr^{-1}) were calculated using the ideal gas law from the linear part of the relationship between N_2O peak area (concentration) and chamber closure time (minutes), and corrected for air temperature, pressure and the ratio of chamber headspace volume to soil surface area using the equation:

$$\mathrm{mgN_2O{-}N\,m^{-2}\,hr^{-1}} = \frac{\alpha \mathrm{VW} * 60}{\mathrm{AMV_{corr}} * 1000} \tag{1}$$

where α is the change in headspace partial pressure during chamber closure period (ppmv/min), V is the headspace volume of the chamber (in L), W is the atomic mass of N in N_2O (28.0), 60 is the conversion from minutes to hours, A is the soil surface area covered by the chamber (m^2), MV_{corr} is the temperature- and pressure-corrected molar volume of N_2O (in L), and 1,000 is the conversion from μg to mg. We conducted regression analysis to determine the effect of irrigation on soil N_2O flux. The regression includes daily precipitation and mean temperature from the LTER weather station archive (https://lter.kbs.msu.edu/datatables/7). Daily irrigation amounts were recorded in the RGE agronomic log (https://lter.kbs.msu.edu/datatables/299). Instead of using these N_2O measurements from one season as the absolute amount by which irrigation increases N_2O, we used the regression to calculate a % increase in N_2O emissions due to irrigation and applied this to 20 years of N_2O flux measurements from the rainfed corn-soybean-wheat no-till plots at the KBS LTER MCSE, reported in Gelfand et al. (2016).

2.5 | Fossil fuel emissions from pumping

The EPA Power Profiler (USEPA, 2017b) provides the % of energy that comes from oil, natural gas, coal, and other energy sources by postal code (KBS postal code is 49060). The US Energy Information Administration (2017) reports source-specific CO_2 emissions per GJ energy. West and Marland (2002) provided an estimate of the energy required to pump groundwater (13.3 GJ ha^{-1} m^{-1} irrigation water), used also in Mosier et al. (2005) and Sainju et al. (2014). We used the % energy from fossil fuels, CO_2 emissions per GJ of fossil fuel energy (specific to each type of fossil fuel), and energy required to pump from the well to the sprinkler at our site to calculate total fossil fuel CO_2 emissions per m of pumped groundwater for irrigation per year.

2.6 | Net irrigation impacts

We calculated net irrigation GWI for two different irrigation scenarios at KBS: 0.18 m, which is the average annual irrigation amount for southwest Michigan (R. Pigg, Michigan Department of Agriculture and Rural Development, pers. comm.), and 0.43 m, which was the amount applied at KBS during the 2012 drought and the maximum applied to date. These irrigation scenarios are hereafter referred to as average and high, respectively. We assume a linear relationship between irrigation amount and process rates. We estimated the impact of irrigation on the total GWI of the KBS LTER no-till cropping system by adding the long-term GWI for the rainfed MCSE no-till system (Gelfand & Robertson, 2015) to the difference between irrigated and rainfed components measured in the RGE in this study. GHG intensity (GHGI, kg CO_2e kg^{-1} yield) was calculated for years 2005–2017. The GHGI is the GWI (including that year's irrigation amount, if applicable) divided by that year's crop yield (kg ha^{-1} $year^{-1}$). GHGIs were averaged by crop with five years of corn and four years each of soybean and wheat.

Multiple linear regression analyses were conducted on the SOC and soil N_2O data in R version 3.4.2 (R Core Team, 2017) using the ggplot2 package for plotting (Wickham, 2009). Residuals were checked for model assumptions (normality, independence, and homoscedasticity). Model selection was conducted using the Bayesian information criterion and R^2.

3 | RESULTS

Mean dissolved CO_2 concentrations in the irrigation wells sampled in the KBS, Kalamazoo, and St. Joseph areas were 27.9 (±1.9), 20.8 (±3.2), and 14.2 (±1.2) mg/L, respectively (Supporting information Table S1), ranging from 2.6 to 73.0 mg/L (1,800–40,600 ppmv). Mean dissolved N_2O-CO_2e concentrations in the KBS, Kalamazoo, and St. Joseph wells were 14.2 (±1.0), 3.7 (±0.9), and 1.5 (±0.4) mg/L, respectively (Supporting information Table S1), ranging from 0 to 47.7 mg/L (0–26,000 ppmv). The GWIs of CO_2 and N_2O degassing from groundwater at KBS in the average and high irrigation scenarios were 7.0 (±0.2) and 16.8 (±0.6) g CO_2e m^{-2} $year^{-1}$, respectively (Figure 2). KBS groundwater had somewhat higher mean concentrations of N_2O and nitrate (9.8 ± 0.2 mg NO_3^--N/L) than the other wells (3.7 ± 1.4 mg NO_3^--N/L; Supporting information Figure S2).

SOC pools in the irrigated plots were significantly greater than in the rainfed plots (Supporting information Table S2). The following model accounted for 79% of the variability in SOC and is significant ($p < 0.001$):

$$\text{kg SOC m}^{-2} = \text{irrigation} * \text{lime} + \text{irrigation} * \text{depth} + \text{N fertilizer} \quad (2)$$

(Figure 3, Table 1). According to the model, the upper 25 cm of irrigated soils had 2.45 (95% CIs: 2.23, 2.67) kg SOC m^{-2} while rainfed soils had 2.15 (1.95, 2.35) kg SOC m^{-2}. Irrigated soils had thus accumulated on average 0.30 kg SOC m^{-2} over 12 years or 25 g SOC m^{-2} $year^{-1}$ more than the rainfed soils. Depth (0–10 cm vs. 10–25 cm), irrigation (yes/no), and lime (yes/no) were significant

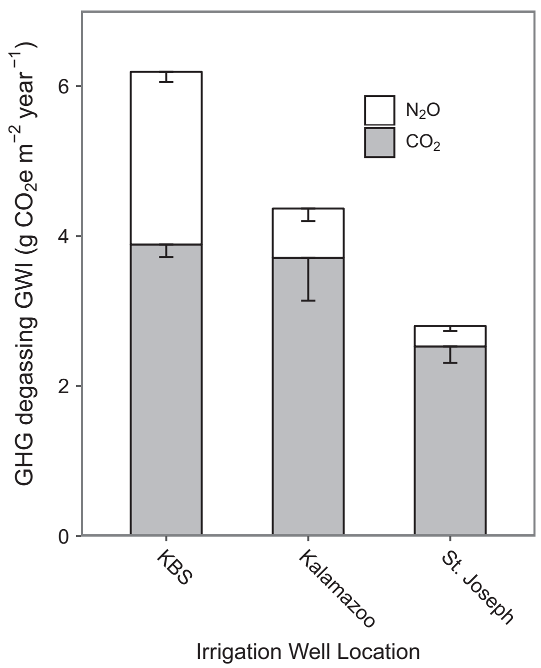

FIGURE 2 Global warming impact (GWI) of greenhouse gases (GHGs) dissolved in groundwater equilibrating with the atmosphere (degassing) in an average irrigation year (0.18 m applied). The KBS bar represents 19 measurements (not including sample replicates) from two irrigation wells at KBS. The Kalamazoo and St. Joseph bars represent the 5 and 17 irrigation wells sampled in the Kalamazoo River and St. Joseph River watersheds, respectively. Error bars represent the standard error of the mean of all replicates per region per gas; only lower SE bar shown to avoid overlapping error bars

TABLE 1 Regression analysis predictors of soil organic C pools (SOC, kg C/m^2) and soil N_2O emissions (g N_2O-N ha^{-1} day^{-1}) at the Resource Gradient Experiment (RGE)

	Estimate	*SE*	*t*-value	Pr(>|t|)	% of R^{2a}	Adj. R^2
SOC Model: kg C m^{-2} = IRR*lime + IRR*depth + N fert. *Ref. levels: rainfed, without lime, depth 0–10 cm, 0 N fert.*				<0.001		0.79
(Intercept)	1.22	0.06	21.1	<0.001	–	
Irrigated	0.42	0.07	6.0	<0.001	3	
With lime	0.34	0.06	5.5	<0.001	3	
Depth 10–25 cm	–0.48	0.06	–8.7	<0.001	83	
N fert. (kg/ha) corn years	–0.0006	0.0003	–2.2	<0.05	2	
Irrigated * Yes limed	–0.33	0.09	–3.9	<0.001	6	
Irrigated * Depth 10–25 cm	–0.20	0.08	–2.6	<0.05	3	
Soil N_2O Model: (N_2O flux)$^{1/3}$ = Wet/dry day + Temp. + IRR + N fert.*Harvest date + N Fert. Date *Ref. levels: dry day, rainfed, 0 N fert., before harvest,>10 days after N fert.*				<0.001		0.32
(Intercept)	1.13	0.08	13.9	<0.001	–	
Wet day	0.40	0.03	11.915	<0.001	26	
Mean daily temp. (°C)	–0.009	0.004	–2.381	<0.05	1	
Irrigated	0.33^b	0.03	10.792	<0.001	21	
N fert. (kg/ha) wheat	0.0030	0.0004	7.106	<0.001	26	
After harvest	0.14	0.06	2.349	<0.05	9	
≤10 days after N fert.	0.55	0.06	9.351	<0.001	14	
N fert.: After harvest	0.002	0.001	3.516	<0.001	2	

[a]Relative importance as a percent of adjusted R^2. [b]The effect of irrigation when back-transformed (cubed) is 0.04 g N_2O-N ha^{-1} day^{-1}.

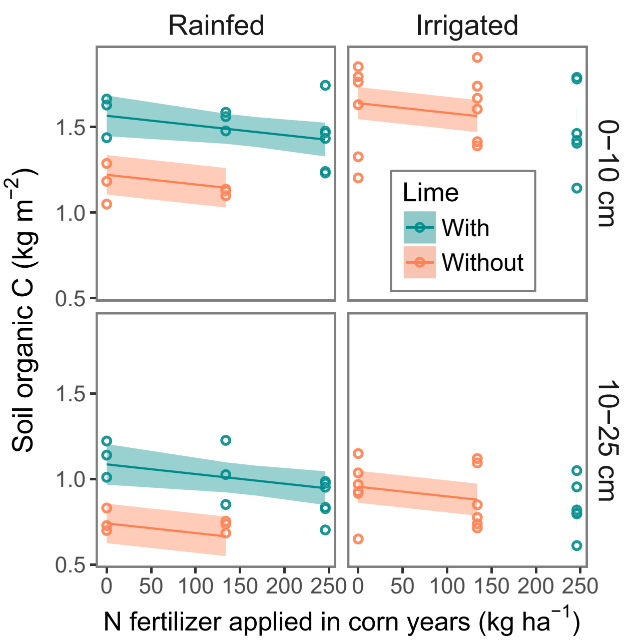

FIGURE 3 Soil organic carbon (SOC) in N fertilization treatments F1, F5, and F8 of the Resource Gradient Experiment. Points are individual cores. Lines and shaded areas represent the model mean predictions and 95% confidence intervals, respectively (Table 1)

categorical variables, while N fertilization rate in corn was a significant continuous variable. The variables lime, irrigation, and N fertilization rate were also significant predictors of soil pH (McGill, 2018). Hence, SOC can be modeled almost as well using simply depth and pH: SOC was positively correlated with pH, and negatively correlated with depth (model R^2 = 0.69 and model p < 0.0001). Predictions of the SOC model (Equation 2) included: (a) SOC pools in the upper rainfed and irrigated soils (0–10 cm) were 0.48 and 0.68 kg/m greater than in the 10–25 cm soils, respectively (i.e., rainfed and irrigated soils accumulated 40 g SOC m^{-2} $year^{-1}$ and 57 g SOC m^{-2} $year^{-1}$ faster, respectively, than 10–25 cm soils); (b) liming increased SOC by 29 g m^{-2} $year^{-1}$ in rainfed plots and by 0.8 g m^{-2} $year^{-1}$ in irrigated plots; and (c) N fertilization in rainfed and irrigated plots reduced SOC by 5 g m^{-2} $year^{-1}$ per 100 kg fertilizer-N/ha. The difference between estimated SOC in irrigated F5 plots and rainfed F5 plots for the total sample depth (0–25 cm) reached 0.30 kg SOC/m^2 (average of limed and unlimed effects) over 12 years (or 25 g SOC m^{-2} $year^{-1}$) and 1.91 m irrigation water. Therefore, the SOC GWI in an average irrigation year at KBS was –8.5 (±5.4) g CO_2e m^{-2} $year^{-1}$ (Table 2).

Soil N_2O flux was significantly higher in the irrigated plots. The following model explained 27% of the variability and is significant (p < 0.001):

$$(\mathrm{N_2O-Ngha^{-1}day^{-1}})^{\frac{1}{3}} = (\mathrm{wetordryday}) + \mathrm{dailymeantemp} + (\mathrm{rainfedorirr.}) + \mathrm{Nfert.rate} * \mathrm{harvest} + (\mathrm{limedornotlimed}) + \mathrm{Nfert.date} \quad (3)$$

(Figure 4, Table 1; Supporting information Table S3). N_2O flux was cube-root transformed to improve its right-skewed distribution. Units are as follows: Measurements were categorized into dry and wet

TABLE 2 Global warming impact (GWI, g CO_2e m^{-2} $year^{-1}$) of irrigation at the Kellogg Biological Station (KBS) in the average and high year scenarios

	Irrigation scenario	
	Average (0.18 m/year)	High (0.43 m/year)
GHG degassing	7.0 (±0.2)	16.8 (±0.6)
SOC	−8.5 (±5.4)	−20.5 (±12.9)
Soil N_2O	28.1 (±7.9)	67.8 (±18.9)
Fossil fuels	14.4	34.7
Net GWI	41.0 (±9.6)	98.8 (±22.9)

Notes. Fossil fuel GWI does not have SE because it is calculated from known values, not repeated measurements. Also see Figure 5.

days depending on whether that plot received 0 or >0 mm, respectively, rainfall and/or irrigation; daily mean temperature is °C; N fertilization rate is kg N/ha; harvest is a factor for before or after July 19, 2017; lime is with or without; fertilization date is a factor grouping the date N fertilizer was applied and the 10 days following it versus all other dates. N_2O flux increased on wet days and with N fertilization rate. The positive relationship with N fertilization rate was steeper after harvest. N_2O emissions decreased at higher mean daily temperatures and were greater in limed plots and during the period of N fertilization. The model predicted that irrigated F6 soil emits significantly more N_2O (13.5 ± 1.0 g N_2O-N ha^{-1} day^{-1}) than rainfed F6 soil (8.7 ± 0.8 g N_2O-N ha^{-1} day^{-1}), an increase of 55%.

We then applied this 55% increase, divided by the 0.138 m of irrigation applied in 2013, to the 20-year average annual N_2O emission rate (3.65 g N_2O ha^{-1} day^{-1} or 39.65 g CO_2e m^{-2} $year^{-1}$) for the three rainfed rotational crops as reported in Gelfand et al. (2016), which means the average irrigation would increase annual N_2O emissions by 28.1 (±7.9) g CO_2e m^{-2} $year^{-1}$.

KBS electricity sources are 1.2% oil, 6.7% non-hydro renewables, 15% natural gas, 16% nuclear, and 60% coal (assumed bituminous; USEPA, 2017b). The fossil fuel CO_2 emissions for the energy required for irrigation were 81.0 g CO_2/m^2 per m of water, which is in agreement with Schlesinger (2000), so for an average irrigation season at KBS the irrigation-associated fossil fuel emissions were 14.4 g CO_2 m^{-2} $year^{-1}$.

The net GWIs of irrigation in the average and high scenarios at KBS were 41.0 (±9.6) and 98.8 (±22.9) g CO_2e m^{-2} $year^{-1}$, respectively (Table 2, Figure 5). The net GWI for the KBS no-till cropping system without irrigation was −13.9 (±30.7) g CO_2e m^{-2} $year^{-1}$ (Table 3, Figure 6; Gelfand & Robertson, 2015), whereas the net GWIs for the irrigated no-till treatment in the average and high year

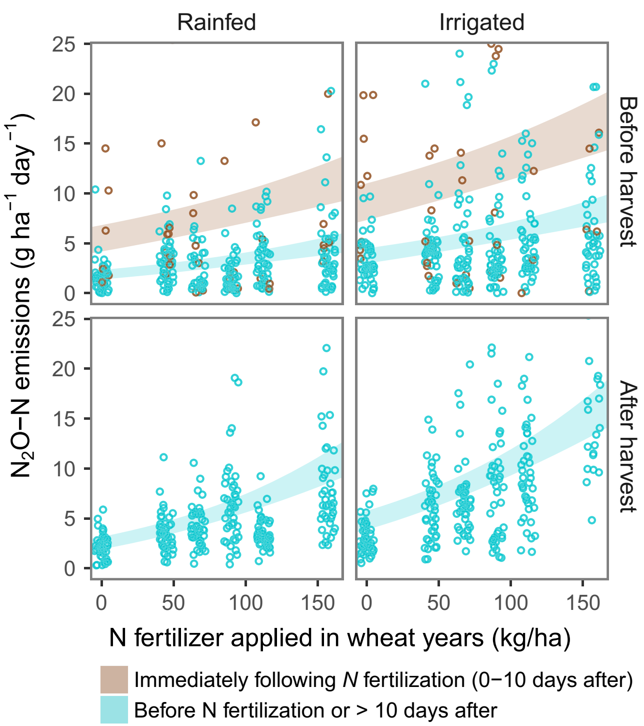

FIGURE 4 Soil N_2O emissions across six N fertilization levels at the Resource Gradient Experiment in 2013, a wheat year in the corn–soybean–wheat rotation. Colors indicate when the measurement was taken relative to the N fertilizer application date. The y-axis is scaled to include the lower 95% of data. Shaded areas indicate the model's 95% confidence interval for the mean (Table 1). Points are jittered on the x-axis to reduce overlap

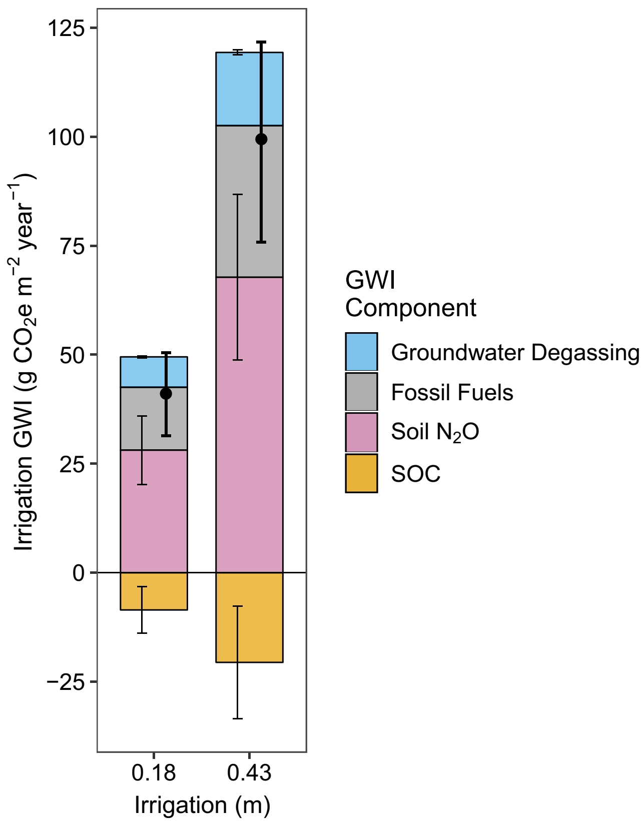

FIGURE 5 Components of irrigation global warming impact (GWI) for average (0.18 m) and high (0.43 m) growing season scenarios. The right-most standard error (SE) bars are for the net GWI of irrigation (black dot); the left-most error bars represent the SE for each GWI component, except fossil fuels, which had no SE (Table 2)

TABLE 3 Global warming impact (GWI, g CO_2e m^{-2} $year^{-1}$) of no-till management in rainfed and average and high irrigation scenarios

	Rainfed	Average (0.18 m/year)	High (0.43 m/year)
Components affected by irrigation			
GHG degassing	0.0	7.0 (±0.2)	16.8 (±0.6)
Fossil fuels	9.0	23.4	43.7
N_2O	39.0 (±3)	67.1 (±8.5)	106.8 (±19.1)
SOC	−122.0 (±30.6)	−130.5 (±31.1)	−142.5 (±32.2)
Component not affected by irrigation			
CH_4	−1.0 (±0)	−1.0 (±0)	−1.0 (±0)
Inputs assumed not affected by irrigation			
N fertilization	33.0	33.0	33.0
Lime	4.0	4.0	4.0
P	0.3	0.3	0.3
K	1.3	1.3	1.3
Seed	7.0	7.0	7.0
Pesticides	15.5	15.5	15.5
Net GWI			
Net	−13.9 (±30.7)	27.1 (±32.2)	84.9 (±38.3)

Notes. Rainfed data are from the no-till treatment at the Long Term Ecological Research (LTER) Main Cropping System Experiment (MCSE; Gelfand & Robertson, 2015; Syswerda et al., 2011). Irrigation values are the rainfed values plus the measured effects of irrigation at the LTER Resource Gradient Experiment F6 N fertilization treatment. N Fert. (N Fertilizer), Lime, P (phosphorus fertilizer), K (potassium fertilizer), Seed, and Pest (Pesticide) refer to the emissions associated with production, transport, and application of those inputs. Values represent rotational averages. Irrigation had no effect on CH_4 emissions. Also see Figure 6.

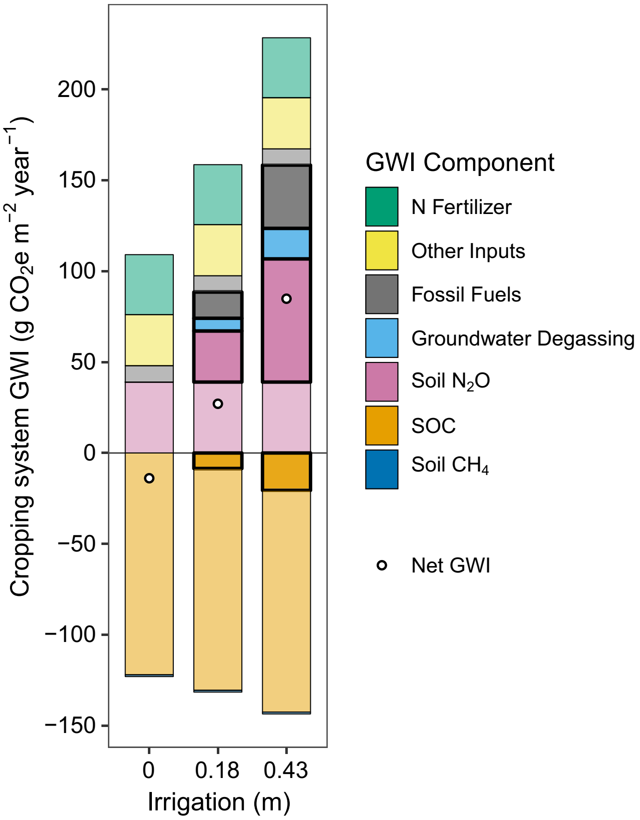

FIGURE 6 Complete global warming impact (GWI) and net GWI for the Kellogg Biological Station's Main Cropping System Experiment (rainfed) and Resource Gradient Experiment (irrigated) no-till cropping systems (rotational average) with three irrigation scenarios: rainfed (0 m), average irrigation (0.18 m), and high irrigation (0.43 m) growing seasons. Components with thicker outlines and darker hues are from Figure 5; components with thinner outlines and lighter hues are from Gelfand and Robertson (2015), which only studied the rainfed systems. The color legend is in the same order top to bottom as components appear in the stacked bars. "Other inputs" includes P and K fertilizers, lime, seeds, and pesticides (Table 3). The amounts of other inputs and N fertilizer were the same in rainfed and irrigated plots at the RGE. Soil CH_4 consumption was −1 g CO_2e m^{-2} $year^{-1}$. See Figure 5 for error bars on measurements from the present study

scenarios were +27.1 (±32.2) and +84.9 (±38.3) g CO_2e m^{-2} $year^{-1}$, respectively.

The net GHG intensity (GHGI, GWI Mg^{-1} yield) of the rainfed and irrigated systems was −0.03 (±0.002) and +0.04 (±0.006) kg CO_2e kg^{-1} yield $year^{-1}$, respectively, as a mean over the crop rotations from 2005 to 2017 (Table 4; Supporting information Figure S3).

4 | DISCUSSION

Groundwater-fed irrigation in this Michigan no-till cropping system was a net source of GHG emissions, despite CO_2e sequestration as SOC. In an average irrigation year, the irrigated system sequestered 7% more CO_2e than the rainfed system (−131.5 g CO_2e m^{-2} $year^{-1}$) but it emitted 45% more CO_2e (158.6 g CO_2e m^{-2} $year^{-1}$; Figure 6). The increase in soil N_2O, fossil fuel, and dissolved GHG emissions due to irrigation alone contributed 17%, 9%, and 4%, respectively, to the system's total emissions in an average irrigation year. Irrigation-driven SOC accumulation added 6.5% to the system's total sinks. The irrigated system was a net source of GHGs under both scenarios investigated: an average year of irrigation (27.1 g CO_2e m^{-2} $year^{-1}$) and a drought year requiring high irrigation (84.9 g CO_2e m^{-2} $year^{-1}$).

4.1 | GHG degassing from groundwater

The degassing of dissolved GHGs from irrigation water contributed 14% of the irrigation-associated GHG emissions at KBS (Figure 5). Measured CO_2 concentrations in the groundwater samples (1,800–40,600 ppmv) were similar to measurements of ~10,000 ppmv from the Southern High Plains aquifer reported by Wood and Petraitis (1984). Our mean CO_2 concentrations varied among the three well locations by a factor of two and mean N_2O concentrations by a factor of eight (Supporting information Table S1). KBS wells tended to have higher N_2O and nitrate concentrations than the other samples (Supporting information Figure S2), but the nitrate concentration (9.8 ± 0.2 mg NO_3^--N/L) is not unusual for groundwater in U.S. agricultural watersheds (Dubrovsky & Hamilton, 2010) where variability

TABLE 4 Mean greenhouse gas intensity at the KBS Resource Gradient Experiment from 2005 to 2017 (five corn years, four years of both soybean and wheat) using each year's irrigation amount (if irrigated) and mean yield of four replicate F6 plots

Crop	Irrigation Treatment	Mean GHG Intensity (kg CO_2e kg^{-1} yield) (±*SE*)		
		Emissions	Sinks	Net
Corn	Rainfed	0.11 (±0.006)	−0.13 (±0.007)	−0.01 (±0.001)
	Irrigated	0.12 (±0.003)	−0.10 (±0.002)	0.03 (±0.003)
Soybean	Rainfed	0.32 (±0.019)	−0.36 (±0.021)	−0.04 (±0.002)
	Irrigated	0.37 (±0.013)	−0.31 (±0.006)	0.07 (±0.016)
Wheat	Rainfed	0.25 (±0.020)	−0.28 (±0.023)	−0.03 (±0.002)
	Irrigated	0.25 (±0.016)	−0.23 (±0.019)	0.02 (±0.006)
Rotational Average	Rainfed	0.21 (±0.015)	−0.24 (±0.016)	−0.03 (±0.002)
	Irrigated	0.24 (±0.016)	−0.20 (±0.014)	0.04 (±0.006)

is affected by land use history, groundwater flow patterns, and the availability of electron donors for biogeochemical reactions (Bohlke, Wanty, Tuttle, Delin, & Landon, 2002). Our dissolved GHG concentrations are well below those measured in headwater streams worldwide—where the majority of dissolved GHGs often result from groundwater inputs (Aufdenkampe et al., 2011; Beaulieu, Arango, Hamilton, & Tank, 2008; Werner, Browne, & Driscoll, 2012)—and are likely to be a conservative representation of GHG degassing in other areas using groundwater for irrigation.

4.2 | SOC sequestration

Over 12 years, irrigation increased the SOC pool in the upper 25 cm of the soil profile by 1% per year, the same rate observed in the A horizon due to no-till at the KBS MCSE (Syswerda et al., 2011). In the Trost et al. (2013) review, the average SOC increase in irrigated compared to nonirrigated cropland in humid climates (one study each in Germany, Austria, Brazil, and Ethiopia) was 2.4% over periods ranging from 8 to 60 years. When standardized by study period the average increase in SOC was about 0.05% per year (range: −0.22%–0.38%). Compared to these studies, the rate of increase in SOC reported here seems consistent.

Furthermore, our results demonstrate a potential link between SOC and soil inorganic C processes—in addition to irrigation, liming increased SOC at KBS (Table 1). Likewise, periodic liming since 1876 at the Rothamsted (UK) Park Grass Experiment increased SOC by 2–20 times compared to unlimed plots (Fornara et al., 2011). Our data for KBS show that SOC and pH are positively correlated (McGill, 2018); perhaps irrigation increased SOC sequestration by increasing crop productivity more than it increased decomposition. The increased SOC might also be partly explained by the application of groundwater alkalinity, which also increases soil pH (McGill, 2018): Fornara et al. (2011) concluded that liming increased soil biological activity and respiration but also increased organic C incorporation into more recalcitrant forms, resulting in a net positive C sink. An additional irrigation treatment with low-alkalinity water could serve to discern the effect of alkalinity versus higher soil water content on SOC storage in the groundwater-fed irrigation plots.

Irrigation and no-till management resulted in similar rates of change in SOC pools at KBS. The F5 irrigated soils had 15% more SOC than rainfed soils after 12 years. At the MCSE, no-till corn–soybean–wheat rotations and fallow fields undergoing ecological succession increased in SOC pools by 13% and 41%, respectively, over 12 years (Syswerda et al., 2011). The eventual steady-state SOC concentrations under irrigation in this humid climate are not known; several of the semi-arid and arid studies included in the review by Trost et al. (2013) showed SOC concentrations and pools in irrigated cropland larger than those under adjacent native vegetation.

4.3 | Greater soil N_2O emissions

Irrigation may have increased N_2O emissions by maintaining wetter soils with anaerobic conditions conducive to denitrification (Bateman & Baggs, 2005; Robertson & Groffman, 2015), responsible for the majority of N_2O emissions at the KBS LTER (Gelfand et al., 2016; Ostrom et al., 2010). Irrigation increased soil N_2O flux per m irrigation water by a factor of four compared to the rainfed system, accounting for 57% of the irrigation-associated GHG emissions at KBS (Figure 5). Soil N_2O emissions contributed the most to switching the no-till system from a net negative GWI when rainfed to a net positive GWI when irrigated (Figure 6). In an average irrigation year, irrigation-associated N_2O emissions exceeded the irrigation SOC credit by threefold (Table 2).

Our soil N_2O emissions model (Equation 3; Figure 4) predicted that N_2O emissions in irrigated plots at the F6 N fertilization rate were 55% greater than those in the F6 rainfed soils. Of three N_2O studies comparing rainfed to irrigated crops, N_2O emissions increased by 55% to 141% (reviewed in Trost et al. (2013)). In a Finnish barley cropping system, Simojoki and Jaakkola (2001) found that N fertilization increased N_2O emissions by nearly fivefold, irrigation of unfertilized soils increased N_2O emissions by sevenfold, and irrigation and N fertilization together increased N_2O emissions by almost tenfold. At the RGE, according to the model (Equation 3), irrigation alone increased N_2O emissions by 0.33 g N_2O-N ha^{-1} day^{-1} and N fertilization at the F4, F6, and F8 levels increased N_2O emissions by 0.20, 0.34, and 0.47 g N_2O-N ha^{-1} day^{-1}, respectively (Table 1). The relative importance of irrigation versus N fertilization rate in determining N_2O emissions likely varies with irrigation and N application rates.

4.4 | Increased fossil fuel emissions

The CO_2 emissions from fossil fuel combustion for energy to pump the groundwater was the second largest source of GWI after increased N_2O emissions, making up 30% of the irrigation-associated emissions. These fuel emissions more than double the total fossil fuel emissions from no-till management at KBS (Gelfand & Robertson, 2015), increasing total fossil fuel emissions from 9 to 23 g CO_2e m^{-2} in an average irrigation year. If all of the fossil fuel energy used at the KBS well came from burning natural gas, the fuel-related emissions from irrigation would drop from 14.4 to 8.9 g CO_2e m^{-2} $year^{-1}$ in an average irrigation year and the no-till system's net GWI would decrease from 19.8 to 14.3 g CO_2e m^{-2} $year^{-1}$, an improvement but not enough to switch the overall GWI from a net source to a net sink. Of course, if the energy source were renewable energy, the fossil fuel emissions would be closer to zero (West and Marland 2002), and the system's net GWI in an average irrigation year would be only 3.7 g CO_2e m^{-2} $year^{-1}$. The fossil fuel emissions associated with irrigation could be more accurately measured with on site pumping and electric usage data.

4.5 | Net GWI of irrigation

In the average irrigation scenario (0.18 m), irrigation-associated emissions were a net source at 41.0 g CO_2e m^{-2} $year^{-1}$ (Table 2): The total GWI of fossil fuel use, soil N_2O emissions, and groundwater degassing exceeded the increased SOC sequestration. The overall GWI of irrigation would be higher in a future hotter, drier climate with greater irrigation demand. In years like our high scenario, for example, which may become more frequent with climate change (Georgakakos et al., 2014; Pryor et al., 2014), we calculated an over twofold increase in the irrigation-associated GWI (Table 2). This assumes a linear rate of change in SOC pool with time and N_2O emissions with irrigation amount. We know from other studies that SOC is a finite pool with a sequestration rate that asymptotically approaches a steady-state balance over time (West & Six, 2007). Although we cannot assess the shape of the curve for SOC accumulation at the RGE, we can estimate a potential steady-state value based on nearby unmanaged soils.

The F5 irrigated plots had 18.7 g C kg^{-1} soil, which is 10.8 g C kg^{-1} soil (or 9.9 kg CO_2e m^{-2} in the upper 0–25 cm) less than the never tilled mid-successional community at KBS on the same soil type (Syswerda et al. 2011). We showed that average annual irrigation increases the SOC pool by 7.9 g CO_2e m^{-2} $year^{-1}$, and Syswerda et al. (2011) demonstrated that no-till management increased SOC by 122 g CO_2e m^{-2} $year^{-1}$, for a total increase of 130 g CO_2e m^{-2} $year^{-1}$ in the irrigated F5 no-till plots. Both of these rates apply to the first 12 years of the RGE and MCSE. If these initial rates continue, it would take irrigated F5 soils at the RGE another 76 years to reach the SOC concentration of the KBS never tilled mid-successional community. Because the rate of SOC increase is likely not linear, it will slow over time, likely taking longer than 76 years.

Additionally, denitrification likely increases exponentially, not linearly, with increasing soil water filled pore space (WFPS) reaching an optimum at 70%–80% WFPS (Bateman & Baggs, 2005; Butterbach-Bahl, Baggs, Dannenmann, Kiese, & Zechmeister-Boltenstern, 2013). In a meta-analysis of agricultural fields across Great Britain, Dobbie and Smith (2003) showed that N_2O emissions increased exponentially with WFPS due to rainfall. Soil type, N fertilization rate, and timing of rainfall and N fertilization interacted with the rate at which N_2O emissions responded to WFPS. Thus, our estimates using all available data may overestimate SOC accumulation and underestimate N_2O emissions as irrigation rate increases, making our estimates of net GWI conservative.

4.6 | Net effects of the irrigated cropping system

Without irrigation, our cropping system was a net GHG sink (–13.9 g CO_2e m^{-2} $year^{-1}$), mainly due to SOC gains under no-till management (Gelfand & Robertson, 2015). That C benefit is effectively undone with irrigation: The same system with irrigation had a net positive GWI (27.1 g CO_2e m^{-2} $year^{-1}$, Table 3) in an average irrigation year. As Midwest summers become hotter and drier (Georgakakos et al., 2014; Pryor et al., 2014), years like our high scenario could become more common, in which case the irrigated system's net GWI would likely increase over threefold compared to the average irrigation year (Table 3). Irrigation-associated emissions made up 30% of the system's total emissions and irrigation-associated sequestration made up 7% of the system's total sequestration. While improving irrigation efficiency could both reduce GHG emissions and minimize groundwater extraction, improving efficiency does not necessarily reduce extraction volumes (Grafton et al., 2018; Pfeiffer & Lin, 2014).

Irrigation increases crop yield, potentially reducing the need for crop production (and resulting GHG emissions) elsewhere (Burney, Davis, & Lobell, 2010; Mueller et al., 2012; Turral et al., 2011). The rotational average of net GHG intensity of our irrigated system was +0.04 (±0.006) kg CO_2e kg^{-1} yield, which is not far from that of the rainfed system (–0.03 (±0.002) kg CO_2e kg^{-1} yield; Table 4). Thus, if used for agricultural intensification in a way that does not increase the relative use of other inputs, irrigation may have little net climate impact. This presupposes, however, that yield increases due to irrigation will reduce pressures to convert land elsewhere to agriculture. If higher yields due to irrigation do not keep land elsewhere from being cultivated, then the improved GHG intensity with irrigation will not benefit the climate. Moreover, irrigation will lead to faster aquifer depletion (Famiglietti, 2014; Gleeson et al., 2012).

4.7 | Including irrigation impacts in GHG inventory methods

In contrast to IPCC GHG reporting methodologies (De Klein et al., 2006), USDA GHG inventory methods (Ogle et al., 2014) account for irrigation's effect on SOC as well as emissions of soil N_2O and fossil fuel CO_2. These three components of irrigation's GWI had the largest absolute values in the present study, suggesting they are the appropriate components to prioritize for inclusion in GHG inventories. But inventories can be improved by accounting for GHG degassing, which made up 14% of irrigation-associated GHG emissions. In order to account for groundwater degassing on a national scale, we need to know the concentration of dissolved GHGs in aquifers used for irrigation and the amount of irrigation water applied. The spatial variability reported here suggests it could be difficult to accurately scale up GHG degassing without local measurements. Irrigation with alkaline groundwater may also affect soil inorganic C reactions (i.e., precipitation or dissolution of $CaCO_3$) differently in other climatic and edaphic settings than what we have observed in humid southwest Michigan.

5 | CONCLUSIONS

Irrigation is a means of adapting to climate change but, as we have demonstrated, can be a net source of net GHG emissions and should be considered in national and global GHG inventories. In this study, GHG degassing from groundwater and increased soil N_2O and fossil fuel CO_2 emissions more than counterbalanced the significantly greater SOC accrual in irrigated cropping systems. As far as we can determine, this is the first study to empirically quantify the GWI for the effect of irrigation on groundwater degassing and soil N_2O emissions relative to a nonirrigated system in the Midwest US. More studies are needed to assess how these impacts vary among locations, cropping systems, and over time. Furthermore, we need to understand the social dimensions of irrigation to better predict how irrigation water use may change under different future climate and economic scenarios. And while agricultural intensification with irrigation may help avoid GHG emissions from land conversion elsewhere, other impacts on water resources will be significant—important considerations in a world with increasing population, food demand, precipitation variability, and groundwater extractions.

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DATA ACCESSIBILITY

Data from this study are archived in the publicly available KBS LTER Data Catalog (https://lter.kbs.msu.edu/datasets/176), and the data and R script are publicly available from the Environmental Data Initiative (https://doi.org/10.6073/pasta/6ab26c6ead42552a1d4f0261e8dac9bc).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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Graphical Abstract

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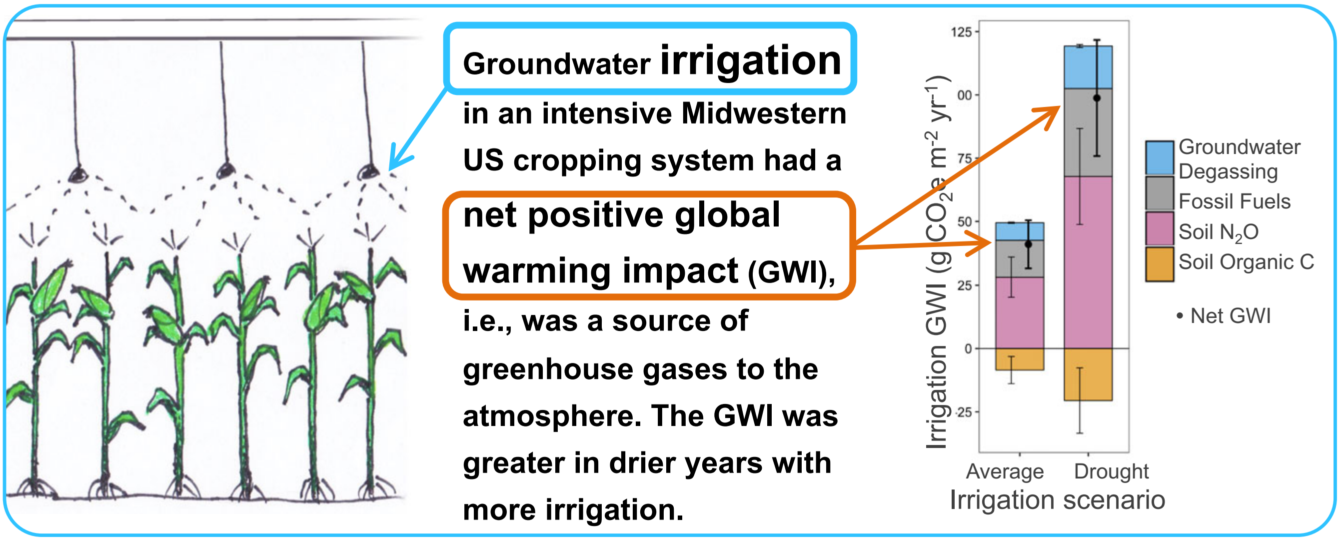

This study compares measurements of the greenhouse gas cost of an irrigated and nonirrigated corn–soybean–wheat system in the Midwest US. Irrigation significantly increased soil organic carbon storage in the upper 25 cm, but not by enough to make up for the CO_2-equivalent costs of fossil fuel power, soil emissions of nitrous oxide (N_2O), and degassing of supersaturated CO_2 and N_2O from the groundwater. Groundwater degassing of CO_2 and N_2O are missing components of previous assessments of the GHG cost of groundwater irrigation; together they were 4% of the irrigated system's total emissions.

Supporting Information

The greenhouse gas cost of agricultural intensification with groundwater irrigation in a Midwest US row cropping system

Bonnie M. McGill, Stephen K. Hamilton, Neville Millar, and G. Philip Robertson

Table S1. Mean (±SE) CO_2 and N_2O concentrations dissolved in groundwater exceeding atmospheric concentrations, i.e., the amount of gas that degasses from the groundwater upon equilibration with the atmosphere.

		CO_2 (mg L^{-1})	N_2O (mg L^{-1})	N_2O as CO_2e (mg L^{-1})	Total CO_2e (mg L^{-1})	GWI (g CO_2e m^{-2} yr^{-1}) Ave-rage year	GWI (g CO_2e m^{-2} yr^{-1}) High year
KBS	Mean	27.9 (±1.9)	0.048 (±0.003)	14.2 (±1.0)	42.1 (±2.9)	7.6	18.1
	Range	10.1 - 73.0	0.007 - 0.160	2.2 - 47.7	-	-	-
Kala-mazoo	Mean	20.8 (±3.2)	0.012 (±0.003)	3.7 (±0.9)	24.5 (±4.1)	4.4	10.5
	Range	4.9 - 40.3	0.0 - 0.034	0.0 - 10.1	-	-	-
St. Joseph	Mean	14.2 (±1.2)	0.005 (±0.001)	1.5 (±0.4)	15.7 (±1.6)	2.8	6.8
	Range	2.6 - 32.0	0.0 - 0.030	0.0 - 9.1	-	-	-

Table S2. Mean Soil Organic Carbon (SOC) at the Resource Gradient Experiment measured in 2016 after 12 years of treatments.

N Fert. treat-ment	Depth (cm)	Irrigation treatment	Mean SOC pool (kg C m^{-2}) (± SE)	Mean SOC concentration (g C kg^{-1} soil) (± SE)
F1	0-10	Rainfed	1.37 (± 0.1)	10.08 (± 0.85)
F1	10-25	Rainfed	0.94 (± 0.09)	7.42 (± 0.73)
F1	0-10	Irrigated	1.57 (± 0.12)	12.05 (± 1.14)
F1	10-25	Irrigated	0.95 (± 0.07)	6.73 (± 0.55)
F5	0-10	Rainfed	1.33 (± 0.1)	10.04 (± 0.77)
F5	10-25	Rainfed	0.88 (± 0.09)	6.7 (± 0.63)
F5	0-10	Irrigated	1.67 (± 0.1)	12.36 (± 0.9)
F5	10-25	Irrigated	0.86 (± 0.09)	6.33 (± 0.74)
F8	0-10	Rainfed	1.43 (± 0.08)	11.12 (± 0.86)
F8	10-25	Rainfed	0.88 (± 0.05)	6.95 (± 0.42)
F8	0-10	Irrigated	1.5 (± 0.1)	11.39 (± 0.79)
F8	10-25	Irrigated	0.84 (± 0.06)	6.58 (± 0.45)

Table S3. Mean N_2O flux at the Resource Gradient Experiment measured in 2013, a wheat year.

N fert. treat-ment	N fert. kg ha^{-1}	Irrigation treatment	Wet day (yes/no)	Before or after harvest	Within 0-10 d after N fert.	Mean N_2O-N flux (g ha^{-1} d^{-1})	Standard error
F1	0	Rainfed	no	before	no	1.62	0.21
F3	45	Rainfed	no	before	no	2.75	0.30
F4	67	Rainfed	no	before	no	2.98	1.17
F5	90	Rainfed	no	before	no	1.46	0.27
F6	112	Rainfed	no	before	no	3.28	0.42
F8	157	Rainfed	no	before	no	3.99	0.78
F1	0	Irrigated	no	before	no	3.08	0.37
F3	45	Irrigated	no	before	no	3.65	0.29
F4	67	Irrigated	no	before	no	5.06	1.57
F5	90	Irrigated	no	before	no	3.71	0.74
F6	112	Irrigated	no	before	no	5.23	1.22
F8	157	Irrigated	no	before	no	9.32	2.03
F1	0	Rainfed	yes	before	no	2.31	0.86
F3	45	Rainfed	yes	before	no	4.20	0.72
F4	67	Rainfed	yes	before	no	8.82	4.36
F5	90	Rainfed	yes	before	no	6.36	3.46
F6	112	Rainfed	yes	before	no	8.43	3.85
F8	157	Rainfed	yes	before	no	8.46	3.76
F1	0	Irrigated	yes	before	no	5.17	1.67
F3	45	Irrigated	yes	before	no	5.44	1.22
F4	67	Irrigated	yes	before	no	13.34	5.38
F5	90	Irrigated	yes	before	no	12.64	5.96
F6	112	Irrigated	yes	before	no	12.86	4.24
F8	157	Irrigated	yes	before	no	15.98	5.83
F1	0	Rainfed	no	after	no	1.97	0.20
F3	45	Rainfed	no	after	no	3.03	0.21
F4	67	Rainfed	no	after	no	3.66	0.27
F5	90	Rainfed	no	after	no	6.14	0.75
F6	112	Rainfed	no	after	no	3.27	0.20
F8	157	Rainfed	no	after	no	9.23	1.58
F1	0	Irrigated	no	after	no	2.77	0.25
F3	45	Irrigated	no	after	no	4.86	0.43
F4	67	Irrigated	no	after	no	5.88	0.47
F5	90	Irrigated	no	after	no	6.36	0.72
F6	112	Irrigated	no	after	no	10.88	1.65
F8	157	Irrigated	no	after	no	14.93	1.77
F1	0	Rainfed	yes	after	no	2.87	0.39
F3	45	Rainfed	yes	after	no	4.98	0.72
F4	67	Rainfed	yes	after	no	9.36	2.62
F5	90	Rainfed	yes	after	no	10.30	3.11
F6	112	Rainfed	yes	after	no	4.41	0.62

Table S3 continued. Mean N_2O flux at the Resource Gradient Experiment measured in 2013, a wheat year.

N fert. treat-ment	N fert. kg ha^{-1}	Irrigation treatment	Wet day (yes/no)	Before or after harvest	0-10 d or > 10 d after N fertilization	Mean N_2O-N flux (g ha^{-1} d^{-1})	Standard error
F8	157	Rainfed	yes	after	no	9.08	1.63
F1	0	Irrigated	yes	after	no	4.57	0.61
F3	45	Irrigated	yes	after	no	8.91	0.96
F4	67	Irrigated	yes	after	no	17.70	7.71
F5	90	Irrigated	yes	after	no	20.53	4.82
F6	112	Irrigated	yes	after	no	18.22	4.93
F8	157	Irrigated	yes	after	no	22.99	3.45
F1	0	Rainfed	no	before	yes	4.08	2.62
F3	45	Rainfed	no	before	yes	9.06	3.13
F4	67	Rainfed	no	before	yes	1.79	0.91
F5	90	Rainfed	no	before	yes	4.08	3.08
F6	112	Rainfed	no	before	yes	2.06	1.13
F8	157	Rainfed	no	before	yes	9.61	4.29
F1	0	Irrigated	no	before	yes	8.73	3.07
F3	45	Irrigated	no	before	yes	7.58	2.74
F4	67	Irrigated	no	before	yes	5.65	2.35
F5	90	Irrigated	no	before	yes	10.82	4.31
F6	112	Irrigated	no	before	yes	12.09	7.36
F8	157	Irrigated	no	before	yes	24.10	15.62
F1	0	Rainfed	yes	before	yes	8.28	2.01
F3	45	Rainfed	yes	before	yes	3.89	2.04
F4	67	Rainfed	yes	before	yes	18.10	9.19
F5	90	Rainfed	yes	before	yes	1.45	NA
F6	112	Rainfed	yes	before	yes	25.99	8.87
F8	157	Rainfed	yes	before	yes	16.84	7.15
F1	0	Irrigated	yes	before	yes	15.68	2.34
F3	45	Irrigated	yes	before	yes	5.34	1.55
F4	67	Irrigated	yes	before	yes	33.04	3.50
F5	90	Irrigated	yes	before	yes	33.11	5.89
F6	112	Irrigated	yes	before	yes	51.70	11.93
F8	157	Irrigated	yes	before	yes	72.80	9.25

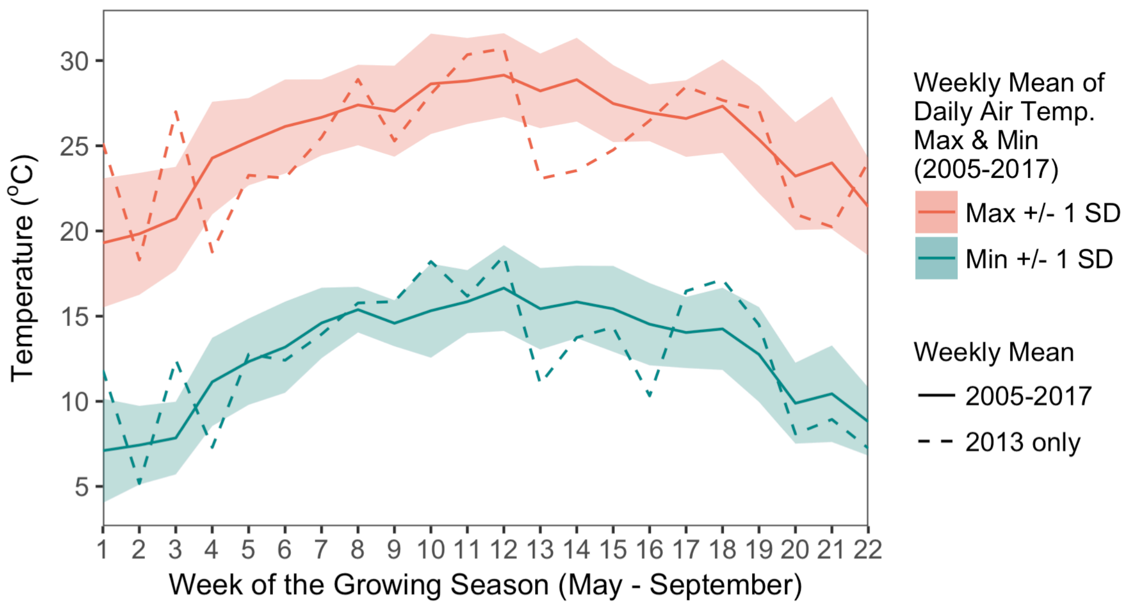

Figure S1. Weekly mean of air temperature maxima and minima during the growing season (May – September) at the Kellogg Biological Station. Soil N_2O emissions were measured during the 2013 growing season. 2013 mean weekly maxima and minima (dashed lines) are, for the most part, within one standard deviation (ribbons) of the 2005 – 2017 means.

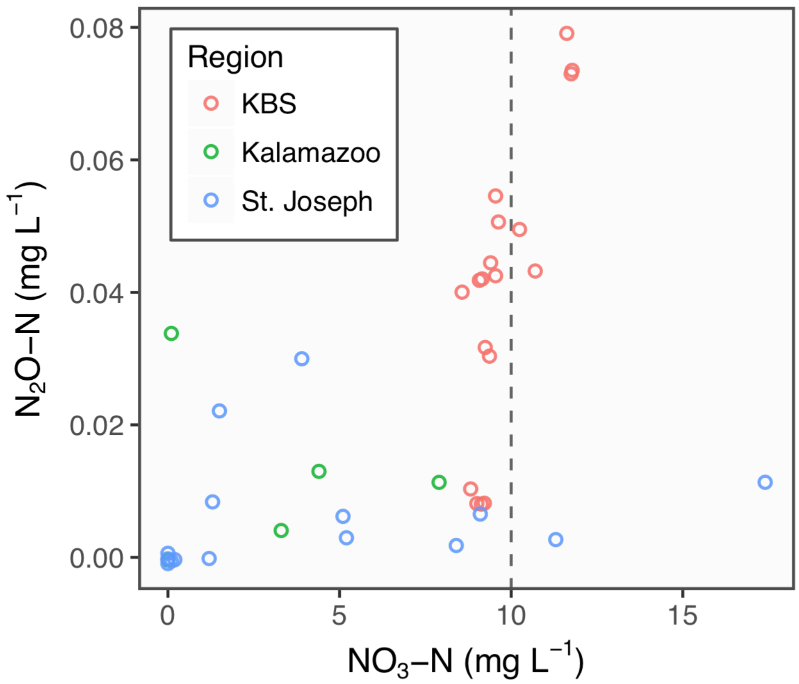

Figure S2. Dissolved N_2O and NO_3^- concentrations in groundwater across SW Michigan sampled in 2016. The dashed line indicates the USEPA water quality standard for NO_3^-.

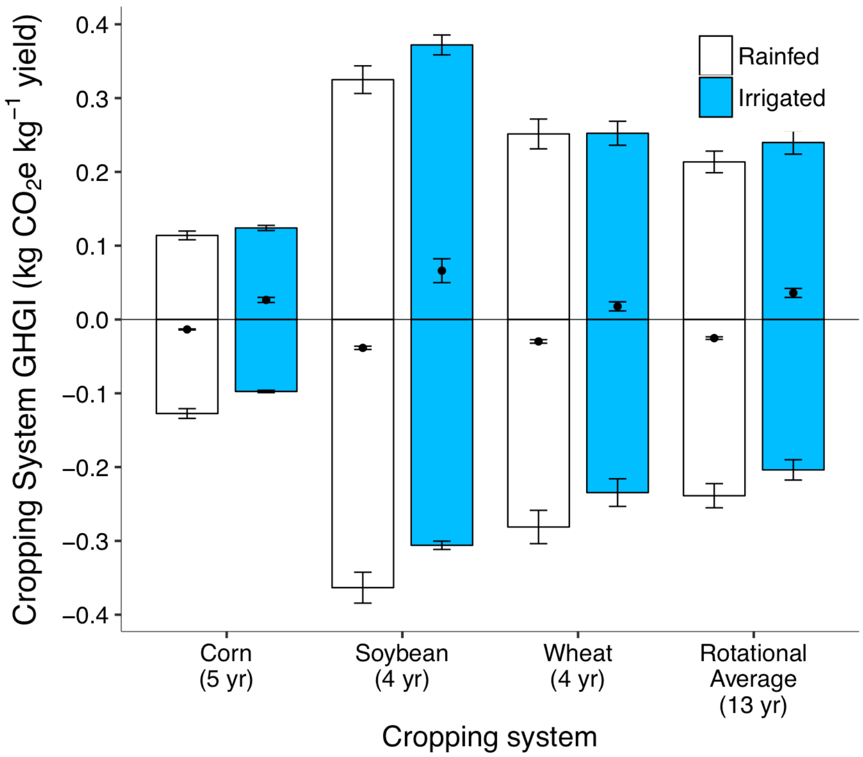

Figure S3. Greenhouse gas intensity (GHGI) per cropping system and irrigation treatment at the Resource Gradient Experiment fertilizer level F6 from 2005-2017. GHGI was calculated using crop yield and global warming impact specific to each year's irrigation amount (for irrigated plots). Bars show the mean positive emissions and negative sinks with standard error (SE) bars. The points are the net of the positive and negative impacts, i.e. the GHGI. Means and SEs were calculated from four replicate plots over the number of years indicated in the x-axis labels.