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Legacy effects of land use on soil nitrous oxide emissions in annual crop and perennial grassland ecosystems

Michael Abraha,[1,2,3,8] Ilya Gelfand,[2,3,7] Stephen K. Hamilton,[2,3,4] Jiquan Chen,[1,2,5] and G. Philip Robertson[2,3,6]

[1]*Center for Global Change and Earth Observations, Michigan State University, East Lansing, Michigan 48823 USA*
[2]*Great Lakes Bioenergy Research Center, Michigan State University, East Lansing, Michigan 48824 USA*
[3]*W.K. Kellogg Biological Station, Michigan State University, Hickory Corners, Michigan 49060 USA*
[4]*Department of Integrative Biology, Michigan State University, East Lansing, Michigan 48824 USA*
[5]*Department of Geography, Environment, and Spatial Sciences, Michigan State University, East Lansing, Michigan 48824 USA*
[6]*Department of Plant, Soil, and Microbial Sciences, Michigan State University, East Lansing, Michigan 48824 USA*

Abstract. Land use conversions into and out of agriculture may influence soil–atmosphere greenhouse gas fluxes for many years. We tested the legacy effects of land use on cumulative soil nitrous oxide (N_2O) fluxes for 5 yr following conversion of 22-yr-old Conservation Reserve Program (CRP) grasslands and conventionally tilled agricultural fields (AGR) to continuous no-till corn, switchgrass, and restored prairie. An unconverted CRP field served as a reference. We assessed the labile soil C pool of the upper 10 cm in 2009 (the conversion year) and in 2014 using short-term soil incubations. We also measured in situ soil N_2O fluxes biweekly from 2009 through 2014 using static chambers except when soils were frozen. The labile C pool was approximately twofold higher in soils previously in CRP than in those formerly in tilled cropland. Five-year cumulative soil N_2O emissions were approximately threefold higher in the corn system on former CRP than on former cropland despite similar fertilization rates (~184 kg $N\cdot ha^{-1}\cdot yr^{-1}$). The lower cumulative emissions from corn on former cropland were similar to emissions from switchgrass that was fertilized less (~57 kg $N\cdot ha^{-1}\cdot yr^{-1}$), regardless of former land use, and lowest emissions were observed from the unfertilized restored prairie and reference systems. Findings support the hypothesis that soil labile carbon levels modulate the response of soil N_2O emissions to nitrogen inputs, with soils higher in labile carbon but otherwise similar, in this case reflecting land use history, responding more strongly to added nitrogen.

Key words: climate change; Conservation Reserve Program; corn; emission factor; grassland; greenhouse gas; labile soil carbon; land use change; no till; restored prairie; smooth brome grass; switchgrass.

Introduction

Land use change (LUC) due to agricultural conversion has greatly modified the Earth's land surface, with ~38% of global land in agricultural production (World Bank 2016). Converting native ecosystems such as grasslands into agriculture increases soil erosion (Montgomery 2007), degrades water quality (Bennett et al. 2001), and leads to habitat and biodiversity loss (Reidsma et al. 2006, Lawler et al. 2014, Werling et al. 2014). In addition, LUC can affect regional and global climate through changes in surface energy balances (e.g., Rotenberg and Yakir 2011), water balances (Pielke et al. 2002, Tian et al. 2017) and greenhouse gas (GHG) emissions (Robertson and Tiedje 1985, Fargione et al. 2008, Gelfand et al. 2011, Ruan and Robertson 2013). In particular, conversion to agriculture may cause a pulse of soil carbon dioxide and nitrous oxide (N_2O) emissions to the atmosphere (Grandy and Robertson 2006, Ruan and Robertson 2013). Globally, agricultural soils are responsible for ~60% of total anthropogenic N_2O emissions (IPCC, 2013).

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[7] Present address: The French Associates Institute for Agriculture and Biotechnology of Drylands, The Jacob Blaustein Institutes for Desert Research, Ben-Gurion University of the Negev, Beer-Sheva 84990 Israel.
[8] E-mail: abraha@msu.edu

Land may also be retired out of agricultural production. For example, about 2% of global agricultural land was abandoned between 1990 and 1992 (World Bank, 2016). In the United States, about 30% (~68 × 10^6 ha) of croplands were abandoned between 1850 and 2000 (Zumkehr and Campbell 2013). More recently, forested land in the United States expanded by ~2.3 × 10^6 ha between 1982 and 2007 (Lawler et al. 2014), while ~15 × 10^6 ha of former agricultural lands were retired into the Conservation Reserve Program (CRP) between 1985 and 2007, to be managed as grasslands and other perennial ecosystems (USDA-FSA 2017). However, with recent crop expansion due to grain biofuel demand (Wright and Wimberly 2013, Mladenoff et al. 2016), the total land area in the CRP had declined to ~10 × 10^6 ha by 2016 (USDA-FSA 2017). All of these changes have direct implications for soil–atmosphere N_2O emissions.

One of the major consequences of LUC into and out of agriculture is the change in the soil organic carbon (C) pool (Post and Kwon 2000), which is affected by changes in management practices (e.g., crop choice, tillage, residue management, and fertilization; West and Post 2002, Christopher and Lal 2007, Loecke and Robertson 2009). The effects of land use legacies on soil N_2O emissions have not often been explored, in part due to insufficient long-term post-LUC soil N_2O flux measurements under contrasting land use histories and management.

Here we investigate the effects of land use legacies, reflected in different labile soil C pools, on soil N_2O fluxes in converted cropping systems with different nitrogen (N) fertilizer input rates to examine the degree to which soil labile carbon might modulate a soil's potential response to N inputs. To this end we examine conversions of CRP grasslands and conventionally tilled croplands (AGR) to no-till agriculture and perennial grasslands. The former CRP and AGR lands differed in their soil organic C and N contents, with the former CRP lands having significantly higher soil C and N than the former AGR lands (Appendix S1: Table S1; Abraha et al. 2016).

Materials and Methods

Site description

The study sites are located within the northeastern part of the Midwest U.S. Corn Belt in southwest Michigan at the Great Lakes Bioenergy Research Center of the Kellogg Biological Station (KBS) Long-Term Ecological Research site (42°24′ N, 85°24′ W, 288 m above sea level). The area has a humid continental temperate climate with mean annual air temperature of 9.9°C and mean total annual precipitation of 1,027 mm (1981–2010; Michigan State Climatologist's Office 2013). Total annual N deposition is ~8.4 kg $N\cdot ha^{-1}\cdot yr^{-1}$ (1989–2010; Millar and Robertson 2015). From May through September, mean air temperature and total precipitation are 19.7°C and 523 mm, respectively (1981–2010). Soils are Typic Hapludalfs, well-drained sandy loams (Thoen 1990, Bhardwaj et al. 2011).

Prior to the conversions, three of the converted sites were managed as CRP grasslands that had been dominated by smooth brome grass (*Bromus inermis*) since 1987 (22 yr prior to the conversion) and another three were managed as conventionally tilled corn (*Zea mays* L.)–soybean (*Glycine max* L.) rotations (AGR) for several decades. All six sites were planted to soybean in 2009. From 2010 onward, three sites from each land use history were converted either to continuous corn, switchgrass (*Panicum virgatum* L.), or restored prairie (see Abraha et al. 2016 for species composition). A seventh field (CRP-Ref) was maintained in smooth brome grass as a reference site (Fig. 1).

Glyphosate herbicide (2.9 kg/ha *N*-(phosphonomethyl) glycine; Syngenta, Greensboro, North Carolina, USA) was applied on day of year (DOY) 125 in 2009 to kill the extant vegetation, especially on the former CRP lands, and to prepare all converted lands for no-till soybean planting. Resulting residue was left in place. All lands were then planted to glyphosate-tolerant soybean with a seed drill on DOY 160/161. Glyphosate was again applied on DOY 184 at the former CRP lands and on DOY 205 at the former AGR lands (Appendix S1: Table S2).

Starting from 2010 through 2014, the AGR-C and CRP-C sites (see Fig. 1 for site names) were fertilized in early spring with phosphorus (P_2O_5) and potash (K_2O) and then planted to no-till corn in late April to early May. Herbicide mix was applied at or a few days following planting and later in the season as needed to further suppress weeds. Fertilizer, in the form of urea ammonium nitrate (28% liquid N: ~184 kg $N\cdot ha^{-1}\cdot yr^{-1}$), was applied by split application at corn planting and by side dressing in June. Corn grain was harvested in October with corn residue left in place (Appendix S1: Table S2).

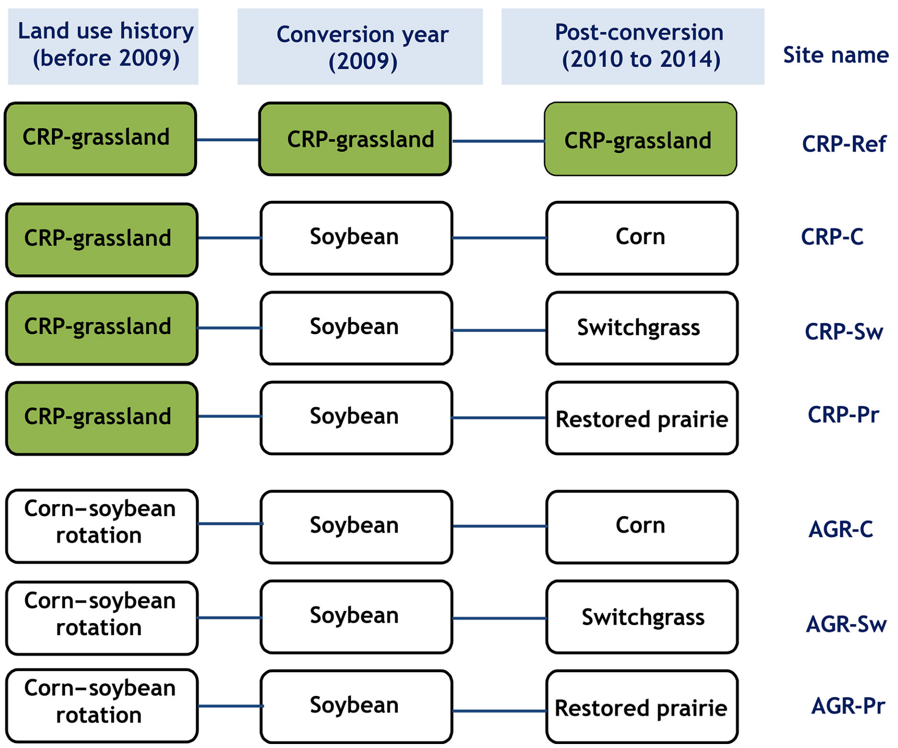

Fig. 1. Schematic representation of the land use history and conversion experiment. The Conservation Reserve Program (CRP) grasslands had been established in 1987 and the conventionally tilled corn–soybean rotation (AGR) fields had been in agriculture since at least 1950. Fields were first converted to no-till soybean in 2009 before conversion to no-till corn (C), switchgrass (Sw), or restored prairie (Pr) in 2010. The CRP-Ref field was kept in smooth brome grass as a reference site.

Switchgrass was planted at the AGR-Sw and CRP-Sw sites at the end of April in 2010. The switchgrass systems were fertilized with urea ammonium nitrate (28% liquid N; ~57 kg $N \cdot ha^{-1} \cdot yr^{-1}$) in all years but 2014. A mixture of 19 native prairie species (Abraha et al. 2016) dominated by C_3 species was planted in early June at the AGR-Pr and CRP-Pr sites. No fertilizer was applied at any time to the restored prairie systems. In the first year (2010), oats were inter-seeded in both the switchgrass and restored prairie grasslands to serve as an over-winter nurse crop. Neither switchgrass nor the restored prairie was harvested in 2010; both were harvested from 2011 onward in early November following autumn senescence. The CRP-Ref site did not receive any agronomic management (Appendix S1: Table S2).

Soil N_2O emissions

Soil N_2O fluxes were measured biweekly from 2010 through 2014 in four locations at each site when soils were not frozen (April–November) using static chambers (Holland et al. 1999) made of stainless steel (KBS LTER 2018). Additional samples were taken following fertilization and precipitation events to capture peak emissions. The sampling interval was increased to once per week during the active growing season following fertilization from 2013 onward. Two consecutive sampling events were missed in 2010, which extended the sampling interval to six weeks for that period. All samples were taken during daytime between 10:00 and 16:00 local time. All sites were sampled on the same day, or rarely on two consecutive days.

Static chambers were cylindrical (28 cm diameter × 23 cm height) and equipped with a vent for pressure equilibration, a gas-tight lid for covering the chamber during sampling, and a septum in the lid for gas sampling. Chambers were placed on the soil surface with about 5 cm inserted into the soil, georeferenced, and removed only for farming activities (e.g., planting, fertilizer and herbicide application, and harvest when applicable). During sampling, the lid was affixed to the chamber and four 10-mL gas samples were transferred into 5.9-mL over-pressurized glass vials (Labco, High Wycombe, UK) at ~15 min intervals using a 10-mL nylon syringe and a 23-gauge needle inserted through the lid septum. The samples were analyzed for N_2O within 2 weeks after collection using a gas chromatograph (Agilent 7890 GC, Santa Clara, California, USA) with a ^{63}Ni electron capture detector (350°C).

Soil C availability

We determined the labile soil C pool using short-term C mineralization assays (Franzluebbers et al. 2000) of soils collected from 0–10 cm depth. We incubated 10 g of sieved (4 mm) and air-dried soil samples (n = 10 sample locations) collected from each system in 2009 and in 2014. We first added deionized water to soil samples to adjust the moisture to 50% water holding capacity. The samples were then incubated in the dark at 25°C for 24 h in 160-mL loosely capped jars, after which jars were flushed with ambient air and capped with gas-tight lids fitted with septa. At ~1 h intervals, four 0.5-mL headspace samples were removed by syringe and injected into an infrared gas analyzer (LI-820 CO_2 analyzer; LI-COR Biosciences, Lincoln, Nebraska, USA) for CO_2 measurement and calculation of fluxes to estimate the labile C pool (mg $C \cdot kg\ soil^{-1} \cdot d^{-1}$) (Robertson et al. 1999).

Data analysis

Cumulative annual N_2O emissions for all systems were determined by integrating daily fluxes, with fluxes for days between samples linearly interpolated. The cumulative annual emissions were then summed to estimate cumulative soil N_2O emissions over the 5-yr study period. Data were analyzed in SAS PROC GLIMMIX (Version 9.4; SAS Institute, Cary, North Carolina, USA). Differences among cropping systems for both annual and overall cumulative N_2O emissions were analyzed by the linear mixed model fit using restricted maximum likelihood (REML), with each site as fixed and chambers within a site as random effects, and years as repeated measurements. The annual and overall cumulative soil fluxes were compared among sites using Tukey's HSD test. Data were checked for normality of residuals and homogeneity of variance assumptions. Soil C mineralization rates among and within systems were analyzed by analysis of variance (ANOVA) and paired t tests, respectively. Where ANOVA was used, comparisons between means were conducted using Tukey's HSD test. Treatment effects were considered statistically significant at $P < 0.05$.

Results

Precipitation and air temperature

The total precipitation during May–September at our study sites, roughly representing the growing season, was 568, 510, 227, 446, and 473 mm from 2010 through 2014, respectively. The growing season mean air temperatures for these years were 19.7°, 19.0°, 19.9°, 18.9°, and 18.1°C, respectively. The growing season precipitation in 2012 (227 mm) was much lower than the 30-yr average (1981–2010) for May through September (523 mm), whereas for the other years growing season precipitation was closer to average. The growing season mean air temperatures for 2010 and 2012 were closer to the long-term average (19.7°C), while the remaining seasons had lower air temperatures than the long-term average.

Soil C mineralization rates

Soil C mineralization rates, representing the labile soil C pool, were significantly higher in soils from the CRP land use history than in soils from the AGR land use history in both 2009 and 2014, except for AGR-Pr in 2014 where rates were statistically similar to those from the former CRP lands (Fig. 2). Rates were generally similar between 2009 and 2014 in a given system with the exception of the AGR-Pr and the AGR-C sites, where rates increased from 51.9 to 85.0 mg $C \cdot kg\ soil^{-1} \cdot d^{-1}$ and decreased from 56.1 to 44.9 mg $C \cdot kg\ soil^{-1} \cdot d^{-1}$, respectively.

Cumulative soil N_2O emissions

Overall cumulative soil N_2O emissions from 2010 through 2014 were highest from the corn and lowest from

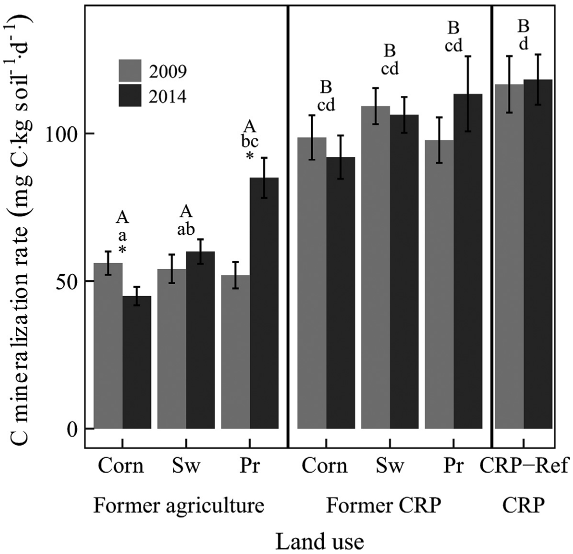

FIG. 2. Short-term soil C mineralization rates (0–10 cm depth) in 2009 and 2014. Lands were converted from the Conservation Reserve Program (CRP) grassland or conventionally tilled corn–soybean rotations (AGR) to no-till soybean in 2009 and then to no-till corn, switchgrass (Sw), and restored-prairie (Pr) systems from 2010 to 2014, while the CRP-Ref site was maintained in smooth brome grass as a CRP grassland. Values are means ± standard errors. Different upper- and lowercase letters indicate significant (n = 10 soil cores per field; $P < 0.05$) differences between mineralization rates across systems in 2009 and 2014, respectively. Asterisks indicate pairwise mean differences of the mineralization rates by t test ($P < 0.05$) within a given system between years 2009 and 2014.

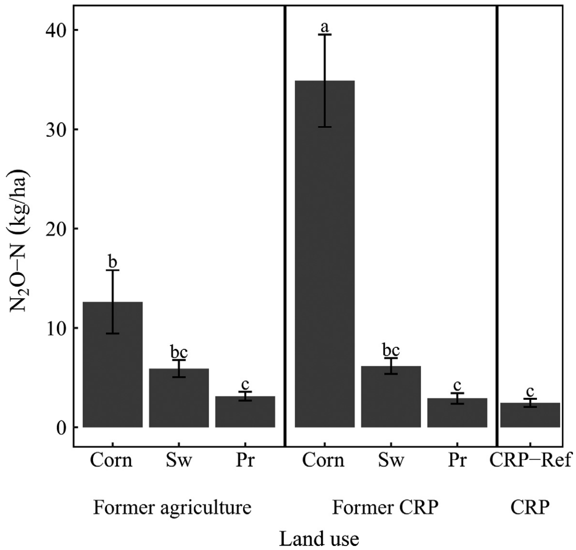

FIG. 3. Cumulative soil N_2O emissions for all systems over the 5-yr post-conversion period (2010–2014). Different letters indicate significant differences in cumulative soil N_2O emissions between systems ($P < 0.05$). Bars and standard errors indicate the cumulative and standard errors of soil N_2O emissions derived from four chambers over the 5-yr period, respectively. See Fig. 1 for LUC and site name information.

unfertilized perennial restored prairies and the CRP-Ref field (Fig. 3). The fertilized corn system with higher labile soil C (CRP-C) had ~3 times higher cumulative emissions (34.9 ± 4.7 kg/ha) than the corn system with lower labile soil C (AGR-C, 12.6 ± 3.2 kg/ha, Fig. 3). The switchgrass systems had similar cumulative emissions (CRP-Sw = 6.2 ± 0.8 and AGR-Sw = 5.9 ± 0.9 kg/ha) irrespective of soil C availability (Fig. 3). The restored prairie systems had similar emissions to the smooth brome grassland at CRP-Ref site (AGR-Pr = 3.1 ± 0.4 kg/ha; CRP-Pr = 2.9 ± 0.5 kg/ha, and CRP-Ref = 2.5 ± 0.4 kg/ha).

While the overall cumulative N_2O emissions for the study duration exhibited clear differences among the cropping systems (Fig. 3), there was a system by year interaction such that individual systems were not consistently different by year (Fig. 4). For example, in the dry year of 2012, all systems emitted a statistically similar amount of N_2O irrespective of N input rates or soil C availability (Fig. 4).

DISCUSSION

Soil N_2O emissions were largely driven by N fertilizer input rate with the magnitude of response to N input rate modulated by labile soil C content, especially at high rates of N inputs. Highest emissions were observed from the corn system with both high N inputs and high labile soil C, which suggests that soil N_2O emissions are modulated by the labile soil C content when N is in adequate supply. The effect of labile soil C could be direct, providing substrates for denitrification, or indirect, for example by enhancing soil water availability in surficial layers (Robertson et al. 2014).

Soil C mineralization rates

Short-term soil C mineralization rates, reflecting labile soil C pools, were approximately twofold higher on the former CRP lands than that on the former AGR lands in both 2009 and 5 yr later in 2014, representing a legacy of the ~22 yr of CRP management (Fig. 2). These findings are similar to previous studies (Post and Kwon 2000, Syswerda et al. 2011, Liu et al. 2016) that showed increased soil C under perennial grassland soils compared to croplands. The consistently high and very similar C mineralization rates observed at the unconverted CRP-Ref site in both years suggest that the differences in labile soil C in the converted systems were due to LUC and management practices after their conversion.

Cumulative soil N_2O emissions

In the corn systems, the CRP-C site, with higher labile soil C content, exhibited approximately threefold higher overall cumulative soil N_2O emissions than did the AGR-C site fertilized at a similar rate (~184 kg $N \cdot ha^{-1} \cdot yr^{-1}$) but with significantly lower labile soil C (Fig. 5). In a study of soil N_2O flux response to rewetting at the same study site, Gelfand et al. (2015) reported a similar difference in N_2O emissions in the corn systems (AGR-C and CRP-C) as in the current study. We also found that soil N_2O emissions were largely influenced by N input rates during the post-LUC period, which is in agreement with many previous studies (Dobbie et al. 1999, Hoben et al. 2011, Smith et al. 2013, Shcherbak et al. 2014, Fig. 5). In the switchgrass systems, both

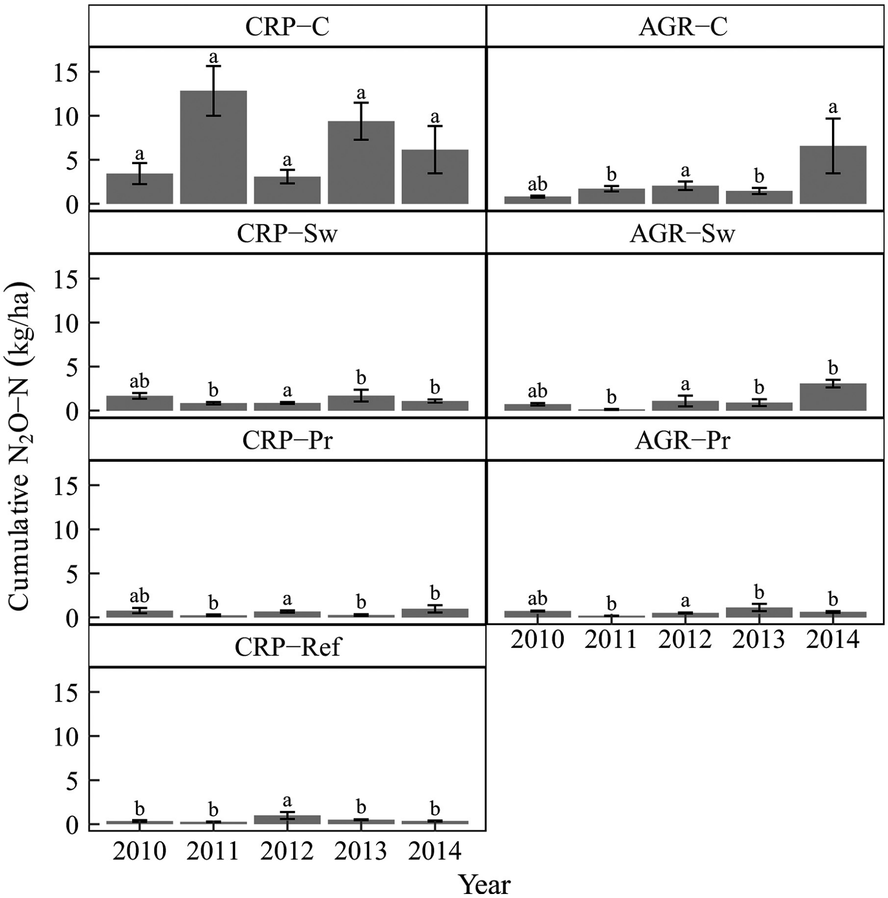

FIG. 4. Annual variation in cumulative soil N_2O emissions for all systems from 2010 through 2014. Different letters indicate significant differences in the cumulative soil N_2O emissions between systems within a given year ($P < 0.05$). Bars and standard errors indicate cumulative and standard errors of soil N_2O emissions derived from four chambers for each year, respectively. See Fig. 1 for land use change (LUC) and site name information.

fertilized at ~57 kg $N \cdot ha^{-1} \cdot yr^{-1}$, the overall cumulative soil N_2O emissions were statistically similar between the two land use histories. We reason this to be a consequence of the large and extensive roots of perennial grasslands and their ability to acquire nitrogen more efficiently (Weih et al. 2011). At the unfertilized restored prairie and smooth brome grass (CRP-Ref) systems, overall cumulative emissions were uniformly low at about half of the emissions from the switchgrass systems (Figs. 3–5). In sum, it appears that soils with similar labile C availability emitted N_2O in proportion to N fertilizer input rates (e.g., CRP-C > CRP-Sw > CRP-Pr ~ CRP-Ref; Figs. 3, 5).

Overall cumulative N_2O emissions exhibited clear differences among systems with different labile C availability and N input rates. However, these differences were observed only for overall cumulative emissions (Fig. 3); totals for any individual year were not consistent (Fig. 4). Inconsistencies were possibly related to inter-annual climatic variability, such as altered precipitation and soil temperatures (Sey et al. 2008). For example, the annual cumulative soil N_2O emissions in the dry year of 2012 were statistically similar in all systems, indicating an overriding influence of soil moisture limitation in the driest conditions. In most other years, however, patterns for annual cumulative emissions reflected the 5-yr sums. In all years, the cumulative fluxes were driven by a few peak emissions that were coincident with fertilizer application and precipitation events (Appendix S1: Fig. S1).

The observed differences in soil N_2O emissions are most likely linked to differences in soil C availability and perenniality. Elevated soil C availability, coupled with N fertilization, would be expected to increase the activity of soil denitrifiers that tend to dominate soil N_2O production in these soils (Ostrom et al. 2010), leading to higher denitrification rates and consequently higher N_2O emissions, as we observed in the corn systems. Similar patterns of higher soil N_2O emissions from soils with higher C availability have been found in soils amended with manure (Velthof et al. 2003) but not in soils with fine particulate organic carbon (Stevenson et al. 2011), pointing to potentially different responses of soil microbes to available carbon sources. Potentially enhanced denitrification rates with higher soil C availability are also supported by the observation that the corn system on former AGR land, with lower labile soil C, had significantly lower N_2O emission than the same crop on former CRP land with higher labile soil C, despite similar N input rates (Fig. 5). On the other hand, overall cumulative N_2O emissions were similar in the fertilized perennial switchgrass despite differences in labile soil C. The higher N acquisition efficiency in root systems of perennial grasslands (Weih et al. 2011) may have maintained low available N for

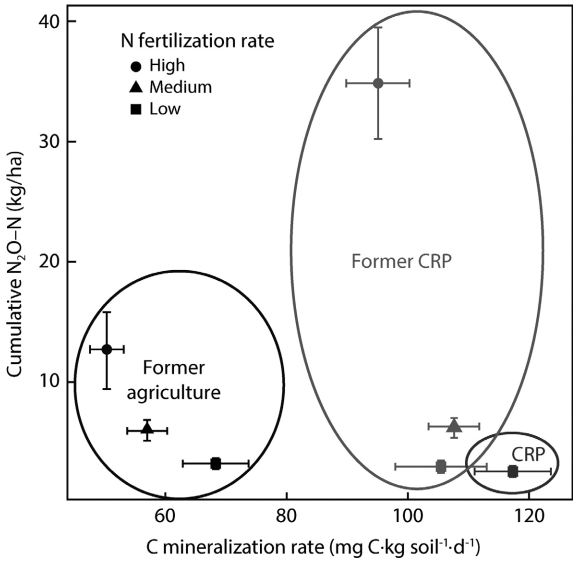

FIG. 5. The relationship between labile soil C content and overall cumulative N_2O emissions (2010–2014) across different N input rates (high, ~184 kg/ha (corn); medium, ~57 kg/ha (switchgrass); and low, atmospheric deposition N inputs only (restored prairie and CRP-Ref). The labile soil C is average for 2009 and 2014. The horizontal and vertical error bars indicate standard errors of labile soil C and cumulative soil N_2O emissions, respectively. The ellipses indicate land use history.

denitrification irrespective of labile soil C content. Similar findings were reported for agricultural soils amended with different levels of C and N in laboratory incubations (Liang et al. 2015), attributed to microbial C and/or N resource limitations, and overall C and N stoichiometry reflecting the C:N ratio (Mooshammer et al. 2014). However, the specific mechanisms of these observed interactions remain unclear.

Our overall average soil N_2O emission rates of 4–57 g·ha^{-1}·d^{-1} from the continuous corn systems fall within the range reported by studies in nearby locations. In a continuous no-till corn system located nearby, fertilized at a rate similar to ours, McSwiney and Robertson (2005) reported an annual average soil N_2O emission rate of ~30 g·ha^{-1}·d^{-1} over three seasons. In conventionally tilled corn at five regional locations, Hoben et al. (2011) found that the average soil N_2O emissions over two seasons ranged 8–60 g·ha^{-1}·d^{-1}, with an aggregate average of 17 g·ha^{-1}·d^{-1}. Interestingly, soil N_2O emissions from Hoben et al. (2011) appear to be directly proportional to the soil organic matter (SOM) content in the top 15-cm soil depth of their study locations. This further supports the linkage between available soil C, N input rates, and N_2O emissions in fertilized cropping systems (assuming labile soil C availability is proportional to total SOM content). The specific mechanisms of the link between the labile soil C pool (mineralizable carbon) and the denitrification potential of soils has been extensively studied (Bijay-Singh et al. 1988, Mitchell et al. 2013) but is still not well understood. Here we demonstrate that soils with higher total and mineralizable C produce higher N_2O emissions than do soils with lower soil C contents fertilized at similar rates. The well-drained soils at our study sites with contrasting total and mineralizable soil C, reflecting land use history, allowed us to explore the effect of available soil C on N_2O emissions without C addition.

Implications for national greenhouse gas (GHG) inventories

National GHG inventories typically assess direct N_2O emissions from agriculture using an emission factor (EF), the proportion of added N emitted as N_2O, assumed to be 1% in the absence of regional evidence otherwise (de Klein et al. 2006). If we assume that N_2O emissions in our unfertilized restored prairie systems (AGR-P and CRP-P) represent background (unfertilized) emission rates in our fertilized AGR and CRP systems, respectively, then we can estimate EFs for our fertilized systems (Appendix S1: Table S3) as 1.0% ± 0.6% (n = 5 yr) and 1.0% ± 0.9% for AGR-C and AGR-Sw, respectively, and 3.5% ± 1.1% and 1.1% ± 0.4% for CRP-C and CRP-Sw.

This calculation suggests first that otherwise identically managed systems can have legacy-dependent EFs, in this case presumably due to labile soil C differences: the EF for CRP-C was 3.5-fold higher than that for AGR-C. Second, that the EFs of corn and switchgrass systems on the AGR land did not differ, despite different fertilizer rates, implies a near-linear relationship between N_2O emitted and the rate of fertilizer input. In contrast, this does not appear to have been the case for the CRP land, where corn had a ~3.2-fold higher EF than switchgrass, implying a non-linear or exponentially increasing response *sensu* Shcherbak et al. (2014). Put another way, the overall change in EF per unit of additional N input (ΔEF, Shcherbak et al. 2014) between switchgrass and corn systems within a given land use history was 0.0004 on the former AGR lands, indicating a close-to linear increase in soil N_2O emission as per the IPCC estimate, but 0.0183 on the former CRP lands, indicating a response much greater than the IPCC estimate. This pattern agrees with Shcherbak et al. (2014) meta-analysis, where they found a stronger soil N_2O emissions response to N fertilizer additions when soil organic C was higher. Our results suggest that national and global N_2O fluxes as computed per IPCC Tier 1 guidelines (de Klein et al. 2006) may especially underestimate soil N_2O emissions for soils with high labile C contents.

Implications for the mitigation of N_2O from cropland ecosystems

The effect of labile soil C availability on soil N_2O emissions found here has significant implications for land management. Reduction of N fertilizer rates in agricultural soils (Millar et al. 2010) and implementation of conservation tillage practices for increasing carbon sequestration (Paustian et al. 2016) have been proposed as effective management tools for mitigation of the negative effects of agriculture on the climate change. Our study suggests that fertilized systems with high labile soil C pools may exhibit higher soil N_2O emissions than those with less labile soil C, which is in agreement with conclusions from recent meta-analyses (van Kessel et al. 2013, Shcherbak et al. 2014). This suggests that N_2O emissions may be more difficult to mitigate in agricultural systems with high labile C stores. Moreover, the interaction is not well represented in process-based soil-vegetation-atmosphere models (e.g., Parton et al. 2001), and may be partly responsible for the uncertainties with which soil N_2O emissions from agriculture are modeled.

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Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.1745/full

Data Availability

Data available from the Dryad Digital Repository: https://doi.org/10.5061/dryad.17g36j4

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Appendix S1

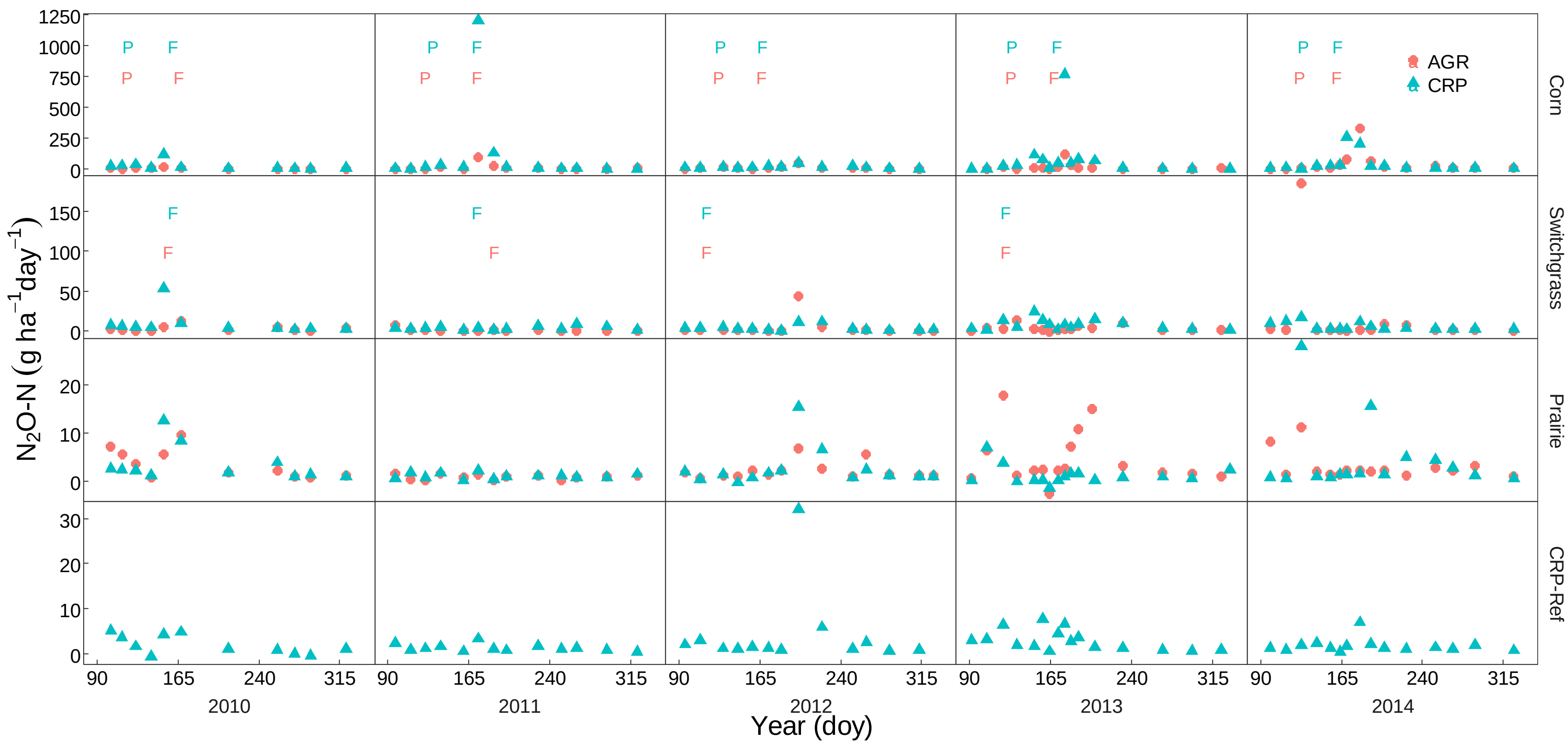

Fig. S1. Average daily soil N_2O emissions from 2010 through 2014 for all systems. Symbols (Red circles for former CRP and blue triangles for former AGR lands) show means from 4 chamber measurements made during April to November. Each panel represents the same crop grown on lands previously managed as Conservation Reserve Program (CRP) grassland and on lands previously managed as conventionally tilled corn-soybean rotation agricultural (AGR) croplands. ‘P' and ‘F’ indicate planting and fertilization events, respectively. Note that the y-axis scales for each crop are different. Error bars are omitted for clarity.

Table S1. Soil physical and chemical properties for the top 0.25 m of the profile at each study site in 2009 before land use conversion: sites were planted to soybean in 2009 and to corn (C), restored prairie (Pr), and switchgrass (Sw) from 2010–2014, converted from either Conservation Reserve Program (CRP) grassland or conventionally tilled corn-soybean rotation agricultural croplands (AGR). The CRP-Ref site was maintained as smooth brome grass as it had been under CRP prior to conversion of the other fields. Means followed by same letter are not significantly different by *t*-test (p<0.05).

Source: soil texture, pH, bulk density, and total carbon and nitrogen – http://lter.kbs.msu.edu/datatables/372.

Site	Soil pH	Bulk Density (g cm^{-3})	Nitrogen (g kg^{-1} soil)	Carbon (g kg^{-1} soil)
CRP-C	6.1^{a}	1.58^{b}	2.0^{d}	21.2^{c}
CRP-Sw	5.9^{a}	1.66^{b}	1.6^{c}	18.5^{c}
CRP-Pr	6.2^{a}	1.59^{b}	1.7^{c}	19.5^{c}
AGR-C	6.4^{a}	1.54^{a}	1.2^{a}	12.2^{b}
AGR-Sw	6.4^{a}	1.79^{c}	1.1^{a}	10.8^{a}
AGR-Pr	5.8^{a}	1.69^{b}	1.4^{b}	13.5^{b}
CRP-Ref	6.2^{a}	1.56^{b}	1.9^{d}	20.9^{c}

Table S2. Timing (Day of Year) of major agronomic management practices performed at each system in each year. Herbicide and fertilizer amount (kg ha^{-1}) applied are included in brackets. Lands were converted from CRP (Conservation Reserve Program) grassland or agricultural croplands (AGR; conventionally tilled corn-soybean rotations) to no-till soybean in 2009 and then to no-till corn (C), switchgrass (Sw) and restored-prairie (Pr) systems from 2010–2014, while the CRP-Ref site was maintained in smooth brome grass. Herbicide mix is a mixture of Lumax, Atrazine 4L, Gramoxone Inteon, $(NH_4)_2SO_4$, States or Cornerstone Plus. Urea ammonium nitrate (28% liquid N)

Year	Activity	AGR-C	AGR-Sw	AGR-Pr	CRP-C	CRP-Pr	CRP-Sw	CRP-Ref
2009	Glyphosate application	146 (2.9)	149 (2.9)	149 (2.9)	125 (2.9)	125 (2.9)	125 (2.9)	–
	Planting	160	160	160	161	161	161	–
	Glyphosate application	205 (2.9)	205 (2.9)	205 (2.9)	184 (2.9)	184 (2.9)	184 (2.9)	–
	Harvest	307	307	306	310	310	309	–
2010	Fertilization							
	• P_2O_5	95 (28)	–	–	–	–	–	–
	• K_2O	95 (67)	–	–	–	–	–	–
	• Urea ammonium nitrate	120 (31)			119 (32)			
	• Urea ammonium nitrate	165 (112)	~155 (57)	–	160 (112)	–	~155 (57)	–
	Planting	118	120	156	119	157	119	–
	Herbicide mix	120	–	–	124	–	–	–
	Harvest	294	–	–	302	–	–	–
2011	Fertilization							
	• P_2O_5 (7-18-37)	104 (59)	–	–	–	–	–	–
	• K_2O (7-18-37)	104 (121)	–	–	–	–	–	–
	• N (7-18-37)	104 (20)	–	–	–	–	–	–
	• Urea ammonium nitrate	125 (34)	–	–	132 (34)	–	–	–
	• Urea ammonium nitrate	172 (134)	188 (57)	–	172 (134)	–	188 (57)	–
	Planting	125	–	–	132	–	–	–

	Herbicide mix	126	–	–	133	–	–	–
	Harvest	307	284	284	311	285	285	–
	Fertilization							
	• P_2O_5	102 (56)	–	–	–	–	–	–
	• K_2O	102 (112)	–	–	102 (85)	–	–	–
	• Urea ammonium nitrate	127 (32)	–	–	129 (37)	–	–	–
2012	• Urea ammonium nitrate	166 (146)	115	–	167 (123)	–	115	–
	• Urea ammonium nitrate	173 (34)	–	–	174 (61)	–	–	–
	Planting	127	–	–	129	–	–	–
	Herbicide mix	127	–	–	129	–	–	–
	Harvest	308	313	313	312	313	313	–
	Fertilization							
	• P_2O_5 (3-14-34-6)	112 (57)	–	–	107 (53)	–	–	–
	• K_2O (3-14-34-6)	112 (138)			107 (128)			
	• N (3-14-34-6)	112 (12)			107 (11)			
2013	• Urea ammonium nitrate	128 (34)	–	–	129 (34)	–	–	
	• Urea ammonium nitrate	168 (179)	123 (57)	–	170 (179)	–	123 (57)	–
	Planting	128	–	–	129	–	–	–
	Herbicide mix	131	–	–	129	–	–	–
	Harvest	307	283	283	305	282	282	–
	Fertilization							
	• P_2O_5 (9-17-0)	126 (6)	–	–	–	–	–	–
	• K_2O (0-0-60 potash)	100 (75)	–	–	100 (112)	–	–	–
2014	• Urea ammonium nitrate	126 (34)			130 (34)			
	• Urea ammonium nitrate	160 (148)	–	–	–	–	–	–
	Planting	126	–	–	130	–	–	–
	Herbicide mix	128	–	–	157	–	–	–
	Harvest	314	296	296	315/316	282	282	–

Table S3. Emission factors (EF) for fertilized corn and switchgrass systems using unfertilized restored prairie systems as control. Lands were converted from CRP (Conservation Reserve Program) grassland or agricultural cropland (AGR; conventionally tilled corn-soybean rotations) to no-till soybean in 2009 and then to no-till corn (C), switchgrass (Sw) and restored-prairie systems from 2010–2014. Soil N2O emissions and N input rates within the same land use history were used for EF computations. Values in parentheses are standard errors of the mean (n = 5 years).

Year	CRP-C	AGR-C	CRP-Sw	AGR-Sw
2010	1.4	0.1	1.6	0.0
2011	6.8	0.8	1.1	-0.1
2012	1.3	0.8	0.3	1.1
2013	5.0	0.2	2.5	-0.4
2014	2.8	3.2	0.2	4.3
Average	3.5(1.1)	1.0(0.6)	1.1(0.4)	1.0(0.9)